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M.Sc. in Conservation Biology

***From cost-effectiveness to economic-efficiency in
conservation planning: the importance of
considering the economic benefits of conservation***

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11 February 2013

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Any success I have achieved in this degree is a result of the grace, love and mercy from my saviour, and friend, Jesus Christ. All my strength, patience and ability to persevere were found in Him.

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Definitions

Protected Area: A conservation area, termed a statutory reserve, which is formally protected and managed under national and provincial legislation in South Africa (Frazee et al. 2003). In this study, 'protected area' and 'reserve' are used interchangeably and are viewed as areas in which natural resource use is prohibited.

Systematic conservation assessment: a technical activity that identifies the location and configuration of priority areas for conservation action based on the principles of representativeness and persistence (Knight et al. 2006).

Systematic conservation planning: the overarching process in which a systematic conservation assessment is combined with a context-specific implementation plan (Knight et al. 2006).

Best solution: The best solution produced by Marxan is the solution that meets all the specified conservation targets with the lowest objective function value (i.e. the most efficient solution) (Ardron et al. 2010).

Summed solution: the summed solution reports the number of times a planning unit was selected as part of a good solution from all runs in a scenario (i.e. indicates selection frequency) (Ardron et al. 2010).

Priority area: refers to a site selected for conservation action based upon their selection frequency and vulnerability to threat.

Priority set: refers to all the priority areas identified from a single scenario.

Cost-efficiency: degree to which target achievement is maximized for a given resource level (Arponen et al. 2010).

Cost effectiveness: measure of true conservation achievement per unit cost (Arponen et al. 2010).

Economic efficiency: the degree to which net economic benefits are maximized per unit cost when ecosystem services are taken into consideration.

Abstract

Providing an economic case for establishing new protected areas and demonstrating how conservation enhances human well-being is becoming necessary to reinforce moral arguments for biodiversity protection. Accordingly; this study aimed to assess whether the spatial distribution of priority areas changes in accordance with gains in economic-efficiency when ecosystem service benefits are explicitly considered. Using the site-selection software Marxan, priority areas for South Africa were identified under four scenarios, two of which incorporated a spatial cost benefit analysis of the opportunity costs and ecosystem service benefits associated with conservation. Additional scenarios assessed how reserve design and costs changed when communal-land agriculture and resource use were weighted to account for the social costs of conservation. Opportunity costs were spatially variable and greatest in regions where mixed commercial farming was practiced. Economic benefits exceeded costs in the western interior, northern regions and along the eastern coastline. These areas contracted when ecosystem service benefits were more conservatively estimated by applying the principle of additionality. There was minimal to moderate spatial agreement between the scenarios ($0.12 < \kappa < 0.55$). When economic benefits were considered, areas that made more economic sense from a conservation management perspective were prioritized. These changes in spatial distribution were associated with gains in economic efficiency. The cost: benefit ratio was *ca.* 10 times greater than when no economic variables were considered (*scenario 1* = 10.44 vs. *scenario 4* = 1.15). When heavy weightings (> 100) were applied, the reserve systems became dispersed ($R = 0.91$ (for weighting factor of 10), $R = 0.52$ (for weighting factor of 1000)) in the former homelands and more costly overall (difference of R2.95B between the extremes). Considering the economic benefits of conservation changes the spatial distribution of priority areas and improves their economic efficiency. Disregarding economic benefits may compromise the implementation potential of priority areas, particularly when the economic benefits of competing land-uses are brought to the table. Furthermore, it is important to explicitly consider the social costs of conservation and consider resettlement or compensation costs among the trade-offs.

Chapter 1: Literature Review – an overview of systematic conservation planning

Conservation

The field of conservation biology emerged in the early 1980s, with the explicit aim of halting and preventing the loss of biodiversity (Sarkar et al. 2006). Along with management, population viability and genetics, a primary focus within this field has been the selection and design of protected areas (Sarkar et al. 2006; Egoh et al. 2007). However, such conservation actions have been constrained because they are typically incompatible with other, more economically beneficial land-uses (Margules & Pressey 2000). As a result, protected areas were historically established in an *ad hoc* manner and concentrated on land that was considered too remote or unproductive to be economically important (Margules & Pressey 2000; Schmitt 2011). The legacy of such an approach is that many protected areas are located in remote places that do not adequately represent the biodiversity of their respective regions (Margules & Pressey 2000; Schmitt 2011). Although conservation priorities have expanded, designing protected areas to protect and sustain biodiversity remains a critical component of conservation (Petrosillo et al. 2010; Rands et al. 2010). Systematic conservation planning evolved in the late 1980's out of an increased interest in going beyond prioritization based upon opportunism, as was typically the case at that time (Cullen 2012).

Systematic conservation planning

Typically, systematic conservation planning produces a conservation plan for establishing protected areas. Fundamentally, conservation plans aim to protect representative biodiversity elements from processes that threaten their existence in the wild (Margules & Pressey 2000). Systematic conservation planning comprises of a systematic conservation assessment and a context-specific implementation plan, and is currently viewed as the most effective tool for identifying and establishing protected area systems (Stewart & Possingham 2002; Egoh et al. 2007). Key concepts that have guided systematic conservation planning are complementarity, representativeness, irreplaceability and vulnerability (Sarkar et al. 2006; Turner et al. 2007). Linked to these concepts are the principles of efficiency, flexibility and persistence (Cullen 2012).

Given limited financial resources and time, the goal of conservation assessments is to identify priority areas from where conservation activities should proceed (Schmitt 2011). Therefore, conservation assessments typically solve a “*constrained optimization problem*”, where area (or cost) is the variable being minimized (optimized) under the constraint that all specified biodiversity targets must be met (Sarkar et al. 2006). Essentially, conservation assessments attempt to solve a “*cost-effectiveness problem*” (Naidoo et al. 2006). Conservation plans can be proactive (prioritize areas of low vulnerability), reactive (prioritize areas of high vulnerability), or representative (where all areas

important for conserving a representative sample of the regions biodiversity are sampled) (Turner et al. 2007; Schmitt 2011). Despite weaknesses in the systematic conservation planning process, the effectiveness of conservation plans comes from their efficiency in allocating limited resources to achieve conservation goals, its defensibility and flexibility in the face of competing land uses, and its accountability in allowing decisions to be critically reviewed (Margules & Pressey 2000).

The “implementation crisis”

A weakness of the systematic conservation planning process is that only a handful of plans developed by conservation biologists have been implemented by organizations and/or individuals who enact land transactions (Newburn et al. 2005; Sarkar et al. 2006). Although prescriptive conservation plans, such as those developed by conservation assessments, may be optimal in theory, they are often difficult or impossible to implement (Knight et al. 2006; Naidoo et al. 2006). This phenomenon is known as the “*implementation crisis*” (Knight et al. 2006: 740). As Margules & Pressey (2000: 250) stress: “*there is a world of difference between the area selection process and making things happen on the ground*”.

The most commonly cited reason for the implementation crisis is that most conservation plans fail to adequately address or incorporate the social, political and economic context of their respective planning regions (Margules & Pressey 2000; Balmford & Cowling 2006; Sarkar et al. 2006; Reyers et al. 2012). In essence, conservation planners have traditionally been unconcerned with practical factors (such as cost) that influence implementation (Newburn et al. 2005; Naidoo et al. 2006). This is despite the fact that conservationists recognize that their efforts are often constrained or adjusted by social, political or economic imperatives (Margules & Pressey 2000; Reyers et al. 2012). The allocation of land to conservation purposes unavoidably competes with the needs of society for settlement, development, agriculture, and the extraction of natural resources (Moore et al. 2004; Cameron et al. 2008). Therefore, such constraints should be explicitly considered in the planning process when implementation is an end goal. Implementation is further complicated by the variety of people and agencies operating within an area, and by the time required to establish conservation management (Margules & Pressey 2000). Factors such as the willingness of private landowners to participate, desires of local interest groups, and short-term government priorities need to be considered because protected areas require the support of government and local communities to survive (Kremen et al. 1999; Newburn et al. 2005). Bearing in mind that these assessment tools are *decision-support* systems and not *decision-making* systems, Balmford & Cowling (2006) urge conservation planners to consider the reality that conservation may be more about the choices and decisions that people make than it is about biodiversity. Not all decisions are made based upon the economic consequences of those decisions, but economic consideration is almost always a contributing factor to the decision-making process.

The economic costs of conservation

Biodiversity loss is seldom the motive for human actions; more often it is an unintended side-effect of decisions made for other reasons (Rands et al. 2010). In many cases, biodiversity loss can be considered an economic externality. To reduce biodiversity loss, conservation needs to be mainstreamed into the everyday decisions made by the private and public sectors of society (Balmford & Cowling 2006). Explicitly considering the economic costs of conservation is a step in the right direction because economic value can convey information to local decision-makers about the relative value of inputs and outputs (Montgomery et al. 1999). Choices, based on prioritization procedures such as conservation assessments, will almost always encounter some type of cost (Cullen 2012). The full cost of conservation is typically comprised of five main components: acquisition, management, damage compensation, transaction and opportunity costs (Adams et al. 2010). Frazee et al. (2003) also considered the cost of building local and institutional capacity. All, or some, of these cost components may be encountered during the implementation phase. For this reason, Linnell et al. (2010) are of the opinion that focusing only on the benefits of (or to) biodiversity, and forgetting the cost of conservation, is naïve.

In many ways, the question of where to locate protected areas is a classic economic problem involving the allocation of a limited budget to maximize a desired goal (Polasky et al. 2001). In conservation context, the goal is some specified level of biodiversity protection. Furthermore, the establishment of a protected area may lead to substantial financial losses for other sectors of society (Linnell et al. 2010). Approaching conservation from this perspective, reserve system design process should be considered a joint economic and biological problem (Richardson et al. 2006). Despite the relevance of an economic approach, economists have rarely been centrally involved in the issue of prioritizing protected areas, or more generally in analyzing and informing conservation policy (Polasky et al. 2001).

Typically, economic considerations are given significantly less attention than biological values in prioritization procedures with few approaches taking into account the relative costs of conserving different areas (Balmford et al. 2000; Carwardine et al. 2008). This is somewhat short-sighted given the significant funding shortages for conservation and the constraints that socio-economic conditions place on protected area establishment (Balmford et al. 2000). The most common approach to conservation assessments has been to minimize the number, or extent, of selected areas, implicitly assuming that land costs are homogenous (Carwardine et al. 2008; Adams et al. 2010). Although the economic costs of conservation began to be incorporated into the prioritization procedure during the late 1990s, effective incorporation of land economics remains relatively rare in the conservation planning literature (Newburn et al. 2005; Sarkar et al. 2006; Polasky 2008; Adams et al. 2010; Cullen 2012). Newburn et al. (2005) reported that only 13% of conservation plans *discussed* the economic

costs of conserving habitat as a component of implementation, while even fewer (9%) explicitly incorporate the costs of land acquisition, conservation easements, or management agreements into prioritization procedures. Where costs have been considered, they are often included *ad hoc* during the implementation phase to evaluate the cost of a plan or alternative plans (Carwardine et al. 2008).

The lack of economic considerations in conservation planning has been attributed to limitations such as 1) the scarcity of data associated with the costs of conservation (Naidoo & Adamowicz 2006), 2) the uncertainty of economic data and the perception that they are unreliable (Naidoo et al. 2006; Arponen et al. 2010; Carwardine et al. 2010; Reyers et al. 2012), 3) a lack of clarity concerning the objectives of the prioritization process (Carwardine et al. 2008), and 4) the possibility that a cost-effective solution may be suboptimal from the perspective of biodiversity conservation (Arponen et al. 2010). Concerns of a suboptimal biodiversity outcome may be unfounded. Carwardine et al. (2010) found that sites of very high or very low conservation value maintained their priority status regardless of the cost of the site. Egoh et al. (2010) also believe that sites with low biodiversity value will not be prioritized simply because they are cheap. Most concern is expressed over the uncertainty of cost data. Data on the costs of conservation are uncertain because, like other economic data, they can vary according to individual perceptions, preferences, values and/or the free market (Arponen et al. 2010; Reyers et al. 2012). Hopefully, in the not too distant future, being able to consider the dynamics of conservation costs will provide a means for dealing with the inherent uncertainty of economic data (Naidoo et al. 2006).

Despite the limitations, cost data are an important component in an efficient conservation decision-making process. Efficiency is crucial because conservation efforts tend to be constrained by existing and future budget shortfalls (Balmford et al. 2000; Reyers et al. 2012), and because conservation competes with other potential land-uses (Ferraro 2002; Sarkar et al. 2006). However, without estimates of cost, claims of “*wise investment*” or “*efficient allocation of effort*” are somewhat unfounded (Murdoch et al. 2007: 376). Improving economic-efficiency requires the explicit inclusion of economic data in the prioritization process, and for conservation planners to grow in their understanding of the costs associated with conservation (Balmford et al. 2000; Naidoo & Ricketts 2006; Knight et al. 2010). Understanding when and how to use the economic component and how much weight it should be given is important (Arponen et al. 2010). Research suggests that costs should be included when 1) costs vary more than biological value (Ferraro 2003), 2) cost and biological value are positively correlated (Ferraro 2003), and 3) the conservation assessment objective is not complementarity or target-based (Naidoo et al. 2006).

Opportunity costs of conservation

As of yet no conservation assessment has been able to fully account for the economic costs of conservation, suggesting that this may be an impossible task. As a result, conservation assessments tend to only consider cost components for which there are adequate data. Together with establishment costs, opportunity costs are one of the major economic barriers to conservation (Bryan et al. 2011). Opportunity costs should be considered in conservation assessments because biodiversity loss is largely due to the perception that opportunity costs of conservation are too high (Balvanera et al. 2001; Singh 2002). Opportunity costs are defined as foregone profits from alternative land-uses such as agriculture, forestry or mining (Margules & Pressey 2000; Blom 2004; Adams et al. 2010). The use of land for biodiversity conservation generally excludes that land from commercial (or other) use; therefore, conservation incurs opportunity costs (Margules & Pressey 2000). Therefore, opportunity costs need to be taken into consideration when designing protected areas (Blom 2004; Adams et al. 2010).

Although quantifying the opportunity costs of achieving a conservation goal is not simple, it is possible to estimate them (Margules & Pressey 2000). For terrestrial protected areas, opportunity costs can be estimated as the value of the land under its next best use, usually agriculture or real estate development (Turpie & Siegfried 1996). Within this general framework, there are three main approaches to estimating opportunity costs of conservation. The first method involves the use of production values to calculate the potential net returns from the most profitable alternative land-use (Osano 2005; Naidoo et al. 2006). Conservation planning studies conducted in a range of countries (e.g. Brazil, Kenya, Uganda, Madagascar and India) have estimated the production value of agriculture (Osano 2005). The second method uses land prices; based on the assumption that the value of a parcel of land is equal to the discounted flow of profit that the parcel is expected to generate into the future (Osano 2005; Naidoo et al. 2006). However, spatially-explicit data on land prices at the necessary resolution are lacking for many parts of the world, in which case they need be modelled (Naidoo & Ricketts 2006). The third option is to estimate opportunity costs using a generalized linear model that is based upon predictor variables of property value such as mean annual precipitation, percentage of untransformed land, vegetation type, and soil type (Osano 2005; Naidoo et al. 2006).

Opportunity costs are also considered an appropriate measure to explicitly account for the social costs of conservation actions (Adams et al. 2010). While opportunity costs associated with conservation may fall on society as a whole, but the burden of these costs is generally felt at the local level by individuals and communities deeply attached to the land and its resources (Linnell et al. 2010). The key to the success of conservation programs is very often held by these local individuals and communities (Linnell et al. 2010). Therefore, regardless of how costs are financed and

distributed, conservation policies will enjoy greater compliance and political acceptability if they attempt to minimize opportunity costs (Chomitz et al. 2005).

Incorporating opportunity costs into systematic conservation assessments

Economic costs have begun to be recognized by the conservation planning community, and useful approaches for integrating the spatial distribution of opportunity costs with layers describing biodiversity benefits in conservation assessments are being developed (Bryan et al. 2011; Reyers et al. 2012). Numerous conservation assessments have explicitly incorporated opportunity costs, with the aim of determining how their inclusion influences reserve design. Following these assessments, several conclusions regarding opportunity costs and area prioritization have been drawn. Firstly, opportunity costs do not correlate with biological value in any simple way (Moore et al. 2004). Secondly, incorporating costs into prioritization processes increases the cost-effectiveness of resulting priority areas (Turpie & Siegfried 1996; Ando et al. 1998; Balmford et al. 2000; Moore et al. 2004; Naidoo & Iwamura 2007). Thirdly, incorporating opportunity costs alters priority areas altogether (Moore et al. 2004; Naidoo & Iwamura 2007). Naidoo & Iwamura (2007) found that priority areas that result from a process including opportunity costs are more geographically dispersed than those that do not. Fourthly, opportunity costs are often more spatially variable than biological data (Polasky 2008). When costs are highly variable, they become a prominent feature in cost-effective conservation, and if the variability in cost data is greater than the variability of biological data then their inclusion becomes even more important (Polasky 2008). Costs at a local level, where systematic conservation assessments are most often applied, usually display less variation than costs at regional or global levels (Arponen et al. 2010). However, local-level studies have demonstrated that the spatial distribution of costs can still be as important as that of biodiversity in determining optimal conservation investments (Naidoo & Iwamura 2007). Fifthly, costs are often highest in areas that are prioritized because the related biodiversity is highly threatened (Osano 2005). All in all, the evidence indicates that opportunity costs can be minimized while still achieving biodiversity targets if they are explicitly considered (Adams et al. 2010).

Studies have, however, highlighted a number of concerns with regard to cost data. Low resolution economic data may cause an underestimation of the profitability of some sites, running the risk of unintentionally including high cost sites in the protected area (Richardson et al. 2006). On the other hand, overestimating the economic value of sites may result in the unintentional exclusion of biologically valuable sites. Therefore, spatial layers of opportunity costs need to cover an extent appropriate for the scale (landscape, regional, catchment) of the assessment (Bryan et al. 2011). Furthermore, layers need to be at a spatial resolution sufficient to capture landscape-scale heterogeneity in both opportunity costs and the biophysical processes that influence both natural and agricultural systems (Bryan et al. 2011). Finally, many studies have also relied on untested

assumptions about factors influencing cost estimates (Adams et al. 2011). While most studies indicate that mapping conservation costs should be given precedence by those “*who wish to achieve the greatest bang for the buck in conservation planning*” (Naidoo & Iwamura 2007: 46), Hughey et al. (2003: 94) reinforced that “*all approaches implicitly or explicitly highlight the difficulties of simultaneous optimization of economic and biological criteria*”.

Economic benefits of conservation

Although efficiency is an important benefit of systematic conservation planning, very few assessments have attempted to achieve economic-efficiency. As Chan et al. (2011) point out, there is an important philosophical difference between cost-effectiveness (cost minimization) and economic efficiency (net economic benefit maximization). When cost-effectiveness is an end goal, economic costs of conservation should be taken into consideration. Similarly, one might expect that when economic-efficiency is an end goal, economic benefits of conservation should be taken into consideration. Questions such as “*do the economic benefits of preventing further losses of biodiversity and ecosystems exceed the costs?*” (Balmford et al. 2011: 162) can only be answered if economic benefits are explicitly traded off against economic costs.

The trade-off

Today we tend to hear less about undoing human damage to nature and more about salvaging nature for economic reasons (Chan et al. 2007). The ecosystem service framework has been promoted as an approach capable of integrating biological, economic and social outcomes in a manner useful for exploring trade-offs in conservation (Wainger et al. 2010; Petrosillo et al. 2010). Simple economic theory can illustrate the trade-off between economic costs and benefits of conservation. As biodiversity and ecosystems decline, the marginal benefits of conservation will exceed the marginal costs of conservation at some point (Balmford et al. 2011). The marginal benefits of conservation tend to increase as biodiverse ecosystems and their services become rarer, while marginal costs (particularly opportunity costs) of conservation tend to decrease as human activities move into less suitable areas (Balmford et al. 2011). The point at which the marginal benefits equal the marginal costs can only be determined if economic measures of both costs and benefits are analysed simultaneously. Furthermore, by including information on biodiversity benefits, economic benefits, and economic costs, conservation assessments can estimate the full return on investment; providing a more accurate prediction of economic viability (McDonald 2009).

Although it is often easier to estimate costs, recent estimates claim that the economic benefits from ecosystem goods and services may be orders of magnitude greater than the cost of maintaining them (Murdoch et al. 2007; Rands et al. 2010). With advances in ecosystem service valuation, the trade-off between the economic costs and benefits of conservation could potentially be addressed in a

cost-benefit analysis (CBA); a familiar economic tool for assessing development projects (Naidoo & Adamowicz 2006). Although CBA has its limitations, it can be useful for clarifying trade-offs and can provide novel insights to conservation planning (Naidoo & Ricketts 2006; Wainger & Mazzotta 2011). Naidoo & Ricketts (2006) list at least three potential insights provided by CBA. Firstly, the spatial distribution of economic benefits and costs could be compared to the distribution of biodiversity, allowing areas of potential synergy or conflict to be identified. Secondly, CBA would highlight areas that have the greatest benefits per unit cost; improving the economic efficiency of conservation actions. Mapping the ratio of costs to benefits across the landscape will clarify the degree to which one exceeds the other for each site (Naidoo & Ricketts 2006). Thirdly, CBA would allow planners to identify areas where conservation makes economic sense, providing an economic case for conservation to reinforce moral and aesthetic arguments. An approach to conservation that includes ecosystem services holds considerable potential for improving biodiversity conservation and human well-being, promising to sustain critical services, open new streams of revenue, and making conservation broad-based and commonplace (Singh 2002; Chan et al. 2006; Goldman et al. 2008).

Ecosystem services

Although there are many elaborate definitions for ‘ecosystem goods and services’, they can be simply defined as ‘*the end products of nature that yield human well-being*’ (Boyd & Banzhaf 2005: 16). The Millennium Ecosystem Assessment (MA) (2005) classified ecosystem services into four categories namely supporting, regulating, provisioning and cultural services. Since the MA there has been widespread interest in the idea of ecosystem services and an increased awareness of their economic value (Naidoo & Ricketts 2006). While, the production and delivery of all ecosystem services requires, to some degree, the presence of living organisms, in some cases the variety of living organisms may not matter as much as the presence of only a few organisms (Van Jaarsveld et al. 2005). This implies that ecosystem services may not always be positively associated with biodiversity, the traditional focus of conservation efforts.

Ecosystem services are not always produced in the area in which they are used; a phenomenon that has contributed toward local populations becoming disconnected from the land stewardship that is required to sustain the provision of services to these populations (Leopold 1949; Levin 1999; Van Jaarsveld et al. 2005). Ultimately, this disconnect from nature has resulted in ecosystem degradation and declines in regulating services (Van Jaarsveld et al. 2005). Preventing further degradation or improving ecosystem condition through successful conservation interventions requires the reconnection of people and nature (Balmford & Cowling 2006). Ecosystem services are, by definition, linked to human well-being; therefore an understanding of and appreciation for ecosystem services may be the point of reconnection. In theory, incorporating ecosystem services into conservation planning could improve the societal relevance of conservation plans by placing shifting the focus to

safeguarding human well-being (Petrosillo et al. 2010). For instance, a cultural ecosystem service, such as recreational opportunity or aesthetic beauty, may, in some instances, be more valuable than biodiversity to those making on-the-ground conservation decisions (McDonald 2009).

Quantifying, valuing and mapping ecosystem services

The concept of ecosystem service conservation is relatively new and much remains to be discovered. Seemingly simple questions such as “*how can the values (ecological, social and economic) of services be mapped to facilitate the use of ecosystem services in spatial landscaping and design?*” (De Groot et al. 2010: 261) still need to be answered. For the economic benefits of ecosystem services to be incorporated into systematic conservation planning, ecosystem service production needs to be quantified, economically valued, and mapped at a relevant spatial resolution (Naidoo & Ricketts 2006). All three of these steps are easier said than done.

In theory, quantifying ecosystem services from an ecological perspective is feasible if researchers can produce an “*ecoservice production function*” that translates ecosystem condition into a measure of ecosystem service provision (Wainger & Mazzotta 2011). The difficulty is that in practice we still lack production functions of this nature (Polasky et al. 2011). Generating functions that produce reliable estimates of ecosystem service provision requires an in depth understanding of how ecosystem structure, composition, and function relate to ecosystem service provisioning. Measuring ecosystem services is further complicated by the fact that their benefits are often difficult to identify, slow to materialize, or dispersed (Chan et al. 2007). Many of these complications are linked to spatial heterogeneity in service provision, which is typically a result of differences in ecological and socio-economic conditions at different scales (De Groot et al. 2010; Chan et al. 2006). Essentially; scale-dependent, spatially explicit models that predict how nature generates and maintains ecosystem services, and how they affects and are affected by people, need to be built (Balmford & Cowling 2006; Chan et al. 2007; Tianhong et al. 2008). Only once the quantity and location of ecosystem service production has been established, these data need to be used to generate maps of ecosystem services value (Naidoo et al. 2008).

Despite progress in developing new valuation techniques, valuing ecosystem services remains complex and has become a major area of research in both environmental and ecological economics (Naidoo & Ricketts 2006). Valuing any service requires some knowledge of the supply of and demand for that service. This in turn requires knowledge of who the beneficiaries are, how they value an individual ecosystem service, and where they live in relation to the area of service production (Naidoo & Ricketts 2006). Consumer preferences for ecosystem services often relate to broad non-use values, such as aesthetic beauty; therefore, valuations cannot be based entirely upon use values (Montgomery et al. 1999). As important as economic valuation is, it is also important to realize that

monetary valuation is currently incapable of capturing the true value of ecosystem services (De Groot et al. 2010). Benefits from ecosystem services vary in the scale of their operation, and tend to accumulate at regional and global scales, rather than at the local scales of service production (Chan et al. 2006; De Groot et al. 2010; Balmford et al. 2011). The mismatch in scale between service production and the experience of service benefits contributes to the problem of mismatched supply and demand. Mismatched supply and demand is further complicated by the disparity between the benefits associated with ecosystem services and the “on-the-ground” outcomes that land managers can observe (Montgomery et al. 1999). Increasing our understanding of the flow of benefits to nearby and distant human populations is important if we are to deal with this spatial mismatch (Naidoo et al. 2008).

As with valuing services, mapping services is much more than simply mapping the ecological function that supports service production. It requires the identification of beneficiaries, their location and use of the service (Egoh et al. 2007). Since ecosystem services are considered in the context of human well-being, demand tends to scale positively with the number of people in close proximity to the area of service production (Singh 2002; Chan et al. 2006). Currently many spatial ecosystem service datasets measure biophysical potential rather than an actual measure of the use of the ecosystem service, limiting the determination of benefits or values of such services (Reyers et al. 2012).

Ecosystem service benefits in systematic conservation assessments

Both scientific and local communities increasingly expect multiple variables to be accounted for in conservation planning (Petrosillo et al. 2010). In line with this, researchers in the conservation planning field are beginning to recognize that including the economic costs and benefits of conservation are important. Economic costs and benefits together are more effective than either type of information alone; therefore, the emphasis should be on identifying, measuring, and valuing a wide range of benefits, costs, and risks to facilitate informed decision-making (Hughey et al. 2003; Arponen et al. 2010). Linnell et al. (2011: 430) encourage renewed interest in the “*inclusion of a serious focus on local benefits and costs in the biodiversity and ecosystem service discourse*”. The question that arises from this is “*how can all the costs and benefits of changes in ecosystem services and values of all stakeholders, be taken into account properly in discounting and cost-effectiveness issues?*” (De Groot et al. 2010: 261). Although ecosystem services are frequently mentioned in conservation assessments, they are rarely included (Egoh et al. 2007). In a review, Egoh et al. (2007) found that 99% of conservation assessments used only species and land class datasets. Of the assessments that did include ecosystem services 63% included cultural services, 50% regulatory services, 44% provisioning services, and 13% supporting services (Egoh et al. 2007). Ecosystem services have been integrated into conservation assessments through 1) biodiversity pattern, 2)

ecological processes, and 3) the mapping of services (Egoh et al. 2007). Only a handful of conservation assessments have gone beyond mapping of services by setting targets for them and prioritizing areas that achieve both ecosystem service and biodiversity targets (Egoh et al. 2007).

There are a number of reasons for the slow incorporation of ecosystem services into conservation assessments. Firstly, at the turn of the century, data describing ecosystem service benefits did not exist, or were incredibly rare (Polasky et al. 2001). Secondly, our ability to characterize the flow of services, in the necessary biophysical and economic terms, at local and regional scales is poor (Chan et al. 2006). Essentially decision makers have imperfect information regarding ecosystem services and their associated value (Chan et al. 2007). However, ecosystem services should not be excluded simply because information relating to them is imperfect. Balmford et al. (2011) suggest that, while some ecosystem services may be undervalued or ignored, the incorporation of any ecosystem service will, in theory, lead to more informed decisions. Thirdly, the spatial mismatch between supply and demand complicates ecosystem-service valuation and the planning for these services (Chan et al. 2006). Fourthly, it is notoriously difficult to determine the contributions that different land-uses contribute to biodiversity targets and ecosystem service values (Reyers et al. 2012). Reyers et al. (2012) found that, even in a data-rich study area, their study was constrained by a lack of information on the impacts of possible land-uses on ecosystem service values.

Despite increased interest, there is no clear approach for explicitly integrating ecosystem services into conservation planning (Chan et al. 2011). Traditionally, the use of ecosystem services has been limited to providing justification for biodiversity conservation, with many studies aiming to establish where trade-offs between biodiversity and ecosystem services occur (Balvanera et al. 2001; Turner et al. 2007). The prioritization debate revolved, for the most part, around an interest in potential synergies between biodiversity conservation and the maintenance of ecosystem services (Egoh et al. 2007; Turner et al. 2007; Chan et al. 2011; Schmitt 2011; Newton et al. 2012). The aim of studies like these is to identify priority areas where conservation will benefit both biodiversity and ecosystem service provisioning. The concern is that areas important for ecosystem service provision might not always be important for biodiversity, and vice versa (Petrosillo et al. 2010; Chan et al. 2011). Several studies have suggested that there is little congruence between areas prioritized for biodiversity and those prioritized for ecosystem services (Chan et al. 2006, 2011; Egoh et al. 2008). This suggests that, in general, it is unlikely that focusing only on conservation priorities will enable the protection of optimal levels of both ecosystem services and biodiversity (Naidoo et al. 2008). In some instances regions important for biodiversity are low in ecosystem service value simply because the original habitat is ecosystem service value poor (e.g. South Africa's Succulent Karoo) (Turner et al. 2007). The extent to which the trade-off between biodiversity and ecosystem services manifests itself does however seem to be somewhat dependent on the objective of the conservation assessment. Egoh et al.

(2010) discovered that relinquishing small amounts of biodiversity resulted in large gains in ecosystem services for the same total opportunity cost in the Little Karoo, South Africa. Turner et al. (2007) found that proactive conservation strategies prioritized areas that harboured a mean ecosystem service value almost three times that of most reactive conservation strategies. This phenomenon was attributed to the fact that ecosystem service values in areas of high vulnerability have already been reduced by habitat loss.

Other studies have revealed that *how* ecosystem services are incorporated into the assessment approach may be more important than drawing conclusions from a spatial trade-off analysis. In a conservation assessment conducted in the Central Interior region of British Columbia, Canada, Chan et al. (2011) found that treating ecosystem services as targeted benefits yielded more spatially cohesive, but more expensive, protected area networks than treating them as co-benefits or costs. The targeted benefits approach treated ecosystem services as “features” in that a set minimum amount of each ecosystem service had to be accounted for in each protected area network generated by Marxan. By contrast, the co-benefit/opportunity cost incorporated ecosystem services in the cost surface of the analysis. The second approach allowed Marxan to maximize the benefits accrued from ecosystem services while minimizing the opportunity costs associated with protected area establishment. They suggest that including the economic value of ecosystem services in the cost surface of the assessment is the simplest means for ensuring that services are given their due weight (Chan et al. 2011). If ecosystem service benefits are incorporated into the cost surface of an assessment, a cost benefit analysis (CBA) approach to conservation becomes possible. Such an approach is important for determining economically optimal conservation actions, yet there are few examples of studies that have examined both the costs and benefits of conservation simultaneously (Naidoo & Adamowicz 2005).

Very few studies have compared the opportunity costs of conservation to the economic benefits of ecosystem services (e.g. Cameron et al. 2008). In a CBA of this nature, the net benefits of conservation equal the sum of the net benefits from the direct-, indirect- and non-use of ecosystem services minus the opportunity costs of conservation (Norton-Griffiths & Southey 1995). In a study conducted in Kenya, Norton-Griffiths & Southey (1995) compared the net return from tourism, forestry and other conservation activities to the opportunity costs of conservation (agriculture and livestock production). In this instance, the opportunity costs of conservation were almost five times greater than the net returns from ecosystem services. In a more recent study Naidoo & Ricketts (2006) compared the opportunity costs and ecosystem service benefits of developing nature corridors in the Mbaracayu Forest Biosphere Reserve, Paraguay. They found that the benefits of ecosystem services were substantial and outweighed the opportunity costs in certain areas. Whether a site passed a spatial CBA was dependent on how many, and which, of the five ecosystem services they valued were

included in the analysis. Although the local benefits did not outweigh the costs for any three of the proposed corridors Naidoo & Ricketts (2006) concluded that a CBA is useful because it indicates the economic shortfall that payment for ecosystem service (PES) schemes would not be able to offset. Newton et al. (2012) assessed the economic viability of establishing ecological networks in the catchment of the River Fome in Dorset, England. They included the market value of arable crop production, livestock production, carbon storage, and timber production as well as the non-market value of flood risk mitigation, aesthetic, recreational, and cultural value. Their conclusion was that the overall market value of an increase in ecosystem service provision resulting from ecological networks is highly dependent on carbon price. Similarly, Polasky et al. (2001) cited the volatility of the carbon price as a major determining factor in the relative cost-effectiveness ranking of different plans.

Estimating the impacts of conservation on ecosystem service delivery

Understanding the relationship between conservation and ecosystem service delivery is a fundamental requirement for incorporating ecosystem services into conservation assessments appropriately (Petrosillo et al. 2010). Statements and questions such as: “*there is a clear need for improving our understanding of how ecosystems change in response to anthropogenic pressures*” (Balmford & Cowling 2006: 692), “*more work is needed on understanding biological benefits of conservation action*” (Murdoch et al. 2007: 376), and “*what is the relationship between ecosystem management state and the provision of ecosystem services?*” (De Groot et al. 2010: 261), highlight the dearth of knowledge regarding this relationship. There is a lack of well-defined approaches for predicting how specific land-use or land-management decisions affect the overall value derived from the landscape (Polasky et al. 2011; Wainger & Mazzotta 2011).

Conceptual frameworks and spatially explicit modelling tools (e.g. InVEST), have provided a means for assessing how ecosystem services are influenced by land management (Nelson et al. 2009; Petrosillo et al. 2010; Balmford et al. 2011; Wainger & Mazzotta 2011; Lester et al. 2013). When modelling biodiversity conservation and ecosystem service outcomes under three alternative land-use scenarios for the Willamette Basin, Nelson et al. (2009) found that all biodiversity and ecosystem service measures were highest under the conservation scenario. In a different study, Polasky et al. (2011) model carbon sequestration, water quality, habitat quality for grassland and forest birds, general terrestrial biodiversity, agricultural and timber production, and the value of land use in urban development, under five alternative land-use scenarios. Their conservation scenario scored well on the species habitat, carbon sequestration and water quality metrics. Although these modelling exercises approach the question from a slightly different angle, they are based upon similar underlying assumptions. The first assumption is that the relationship between ecosystem properties and services is known (De Groot et al. 2010). The second is that land-use and land management decisions influence the ecosystem properties, processes and components that form the basis for service

provision (De Groot et al. 2010; Polasky et al. 2011). The third assumption is that it is possible to identify and quantify the ways in which changes in ecosystem services due to land management affect human well-being (Balmford et al. 2011). The fourth is that ecosystem services interact with one another such that changes in land use or management will increase the provision and value of some services and decrease others (Polasky et al. 2011). Understanding ecosystem service interactions is important because actions taken to deliver one service may inhibit or divert scarce resources away from actions that could have been taken to deliver other services (Lester et al. 2013). The cost of lost provisions from one service due to use of another service depends on the strength and nature of their interaction (Lester et al. 2013).

While these assumptions seem sound, there is scant empirical evidence of the quantitative relationship between land management and ecosystem service provision (Petrosillo et al. 2010). Only a handful of studies have attempted to quantify this relationship. With the aim of assessing the ‘conservation effect’ on the maintenance of ecosystem services in three natural parks in southern Italy, Petrosillo et al. (2010) found that the protected areas as a whole currently provide the same natural capital flow as they did about 50 years ago. Furthermore, they suggest that ecosystem service provision may not increase significantly if there is an affiliated increase in the land-use that (theoretically) provides a greater level of ecosystem service provision (Petrosillo et al. 2010). Essentially, suggesting that the relationship between land-use or management and ecosystem service provision is not necessarily linear. Turpie et al. (2002) found that many South African estuaries would increase in value, due to increases in ecosystem provision, by up to 50% under conservation management. Similarly, Reyers et al. (2012) found that, in the Succulent Karoo and spekboom-dominated thicket habitats of South Africa, formal conservation increases the ability of the land to sequester carbon. These three above mentioned studies provide empirical evidence to suggest that conservation-oriented decisions yield greater ecosystem service provision.

The issue of ecosystem service interactions is an important one. Raudsepp-Hearne et al. (2010) investigated the extent of interactions between ecosystem services in 137 municipalities in Quebec, Canada. Their results indicate that the dominant management goal, such as crop production or enhancing cultural ecosystem services, produce very different bundles of ecosystem services at the municipal level. Actions taken to enhance the supply of some ecosystem services, mainly provisioning services, have led to decreases in many other ecosystem services, including regulating and cultural services (Raudsepp-Hearne et al. 2010). Essentially, because of ecosystem service interactions, it is not possible to optimize the full range of ecosystem services provided by a landscape. Therefore activities that realize the benefits of certain ecosystem services will frequently be at odds with biodiversity conservation, and vice versa (Chan et al. 2011). However, even in cases

of incompatibility, there may be great gains in conservation efficiency by including incompatible services in conservation planning as opportunity costs (Chan et al. 2011).

Study motivations and research questions

From a holistic point of view this study aims to determine whether it is possible for economic efficiency to become the benchmark for “efficiency” in conservation planning without compromising biodiversity protection, and address the issue of developing approaches for integrating ecosystem services into systematic conservation assessments. The primary objectives of this study were to: (1) estimate the spatial variation in economic output of agricultural practices across South Africa, (2) identify conservation priority areas from reserve systems generated by the four scenarios and (3), compare the identified priority areas in terms of area coverage, economic efficiency, and their ability to meet biodiversity targets. As such, the research questions addressed in this study were: (1) how does explicitly incorporating spatial cost benefit data into the cost surface influence the spatial distribution of priority conservation areas, (2) what are the potential costs of failing to explicitly take ecosystem services into account and (3), how does weighting the economic data to account for the social costs of conservation influence the spatial distribution and overall cost of conservation priority areas?

Chapter 2: From cost-effectiveness to economic efficiency in conservation planning: the importance of considering the economic benefits of conservation

Introduction

A traditional focus within the field of conservation biology has been the selection and design of reserve networks to protect and sustain biodiversity (Sarkar et al. 2006; Egoh et al. 2007). However, such conservation actions are often constrained because protected areas are typically incompatible with other, more economically beneficial land-uses (Margules & Pressey 2000). As a result, reserves were traditionally established in an *ad hoc* manner and concentrated on land that was considered too remote or unproductive to be economically important (Margules & Pressey 2000; Schloss et al. 2011; Schmitt 2011). The consequence of such decisions is that many existing reserve networks do not protect a representative sample of their regions' biodiversity.

Increased interest in going beyond establishing reserves based upon opportunism led to the development of systematic conservation planning in the late 1980s (Cullen 2012). Today systematic conservation planning is viewed as the most appropriate and defensible method for designing conservation plans that aim to protect representative biodiversity elements from processes that threaten their existence in the wild (Margules & Pressey 2000). In systematic conservation planning priority conservation areas are identified by a systematic conservation assessment and combined with a context-specific implementation plan to produce a conservation plan (Stewart & Possingham 2002; Egoh et al. 2007). Conservation assessments typically solve a “*constrained optimization problem*”, where area is the variable minimized (optimized) under the constraint that all specified biodiversity targets must be met (Sarkar et al. 2006). The most widely used site-selection software is Marxan (Possingham et al. 2000), which uses a simulated annealing algorithm to generate effective reserve networks.

While systematic conservation planning may be ideal in theory, the issue of the “*implementation crisis*” suggests that conservation plans need to be more reflective of real world constraints that bound implementation procedures and operations (Knight et al. 2006). Only a handful of plans developed by conservation biologists have been implemented by organizations and/or individuals who perform land transactions (Newburn et al. 2005; Sarkar et al. 2006). As Margules & Pressey (2000: 250) stress: “*there is a world of difference between the area selection process and making things happen on the ground*”. The most commonly cited reason for the implementation crisis is that most conservation plans fail to adequately address or incorporate the social, political and economic context of their planning regions (Margules & Pressey 2000; Balmford & Cowling 2006;

Sarkar et al. 2006; Reyers et al. 2012). Conservation efforts are often constrained or adjusted by economic and socio-political imperatives (Margules & Pressey 2000; Reyers et al. 2012). Accounting for these constraints involves integrating both conservation and economic objectives into the conservation planning process (Schneider et al. 2011).

The question of where protected areas should be situated is a classic economic problem involving the allocation of a limited budget to maximize a desired goal; essentially an attempt to solve a “*cost-effectiveness problem*” (Polasky et al. 2001; Naidoo et al. 2006). In a conservation context, the goal is meeting some specified level of biodiversity protection. From this perspective, protected area design is considered a joint economic and biological problem (Richardson et al. 2006). The cost of conservation typically comprises five components: acquisition, management, damage compensation, transaction, and opportunity costs (Naidoo et al. 2006). Historically the aim of conservation assessments has been to minimize the number or extent of selected areas; implicitly assuming that land costs are homogenous (Carwardine et al. 2008; Adams et al. 2010). This assumption is however, more often than not, a poor one. Although the economic costs of conservation began to be incorporated into prioritization approaches in the late 1990s, effective incorporation of land economics remains relatively rare in the conservation planning literature (Newburn et al. 2005; Sarkar et al. 2006; Polasky 2008; Adams et al. 2010; Cullen 2012). This is despite repeated demonstrations of cost savings in achieving conservation goals when economic costs are explicitly considered (Ando et al. 1998; Polasky et al. 2001; Ferraro 2003; Naidoo & Iwamura 2007). The outcomes of these studies were viewed as efficient because they were cost-effective (biodiversity benefits per unit cost were maximized). “*Efficiency*” is a term often used in conservation planning, however, in economics the terms efficiency and cost-effectiveness are not the same. Economic efficiency is maximizing the net economic benefits of a plan, while cost-effectiveness is concerned with cost minimization. As Chan et al. (2011) point out, there is an important philosophical difference between cost-effectiveness and economic efficiency.

In a world where land and monetary resources are limited, it is becoming increasingly difficult to justify biodiversity conservation without also demonstrating benefits for people (Egoh et al. 2007; Chan et al. 2011). Ecosystem services are, by definition, the link between people and nature. Ecosystem services can be simply defined as ‘*the end products of nature that yield human well-being*’ (Boyd & Banzhaf 2005: 16). Therefore incorporating ecosystem services into conservation planning has the potential to improve the societal relevance of conservation (Petrosillo et al. 2010). An approach to conservation that includes ecosystem services holds considerable potential for improving both biodiversity conservation and human well-being by sustaining critical services, opening new revenue streams, and making conservation broad-based and commonplace (Singh 2002; Chan et al. 2006; Goldman et al. 2008).

The question of *how* ecosystem services can be appropriately incorporated into systematic conservation planning remains to be answered. There has been much interest in potential synergies between biodiversity conservation and the maintenance of ecosystem services (Chan et al. 2011; Schmitt 2011). Therefore, the trade-off between ecosystem services and biodiversity has been investigated in an attempt to prioritize areas where conservation will benefit both ecosystem services and biodiversity (Egoh 2009). However, several studies have pointed to generally low correlations between areas prioritized for biodiversity and those prioritized for ecosystem services (Chan et al. 2006; Egoh et al. 2008; Chan et al. 2011). More recently, interests in integrating the economic benefits of ecosystem services into conservation planning have been expressed (Naidoo & Ricketts 2006; Chan et al. 2011). In a novel conservation assessment conducted in the Central Interior region of British Columbia, Canada, Chan et al. (2011) found that treating ecosystem services as targeted benefits yielded more spatially cohesive, but more expensive, protected area networks than treating them as co-benefits or opportunity costs. The targeted benefits approach treated ecosystem services as “features” in that a set minimum amount of each ecosystem service had to be accounted for in each protected area network generated by Marxan. By contrast, the co-benefit/opportunity cost incorporated ecosystem services in the cost surface of the analysis. The second approach allowed Marxan to maximize the benefits accrued from ecosystem services while minimizing the opportunity costs associated with protected area establishment. Provisioning services were considered to be opportunity costs, while all other ecosystem services were considered co-benefits. Their conclusion was that incorporating the economic benefits of ecosystem services into the cost surface of a conservation assessment may be the simplest means for ensuring that ecosystem services are given their due weight in biodiversity conservation.

Following on from the conclusion drawn by Chan et al. (2011), this study applied the approach of incorporating spatial explicit opportunity costs and economic benefits of ecosystem services into the cost surface of the conservation assessments. Opportunity costs are defined as foregone profits from alternative land-uses such as forestry, mining or agriculture (Margules & Pressey 2000; Blom 2004; Adams et al. 2010). In South Africa agriculture is the predominant land-use and is, in many cases, the greatest threat to biodiversity. Therefore, the opportunity costs were given as the estimated economic output from current agricultural practices and natural resource utilization (provisioning ecosystem service) across South Africa. While provisioning ecosystem services are typically opportunity costs of conservation; the inevitable decline in value of regulating and supporting ecosystem services from unconserved land could be considered the opportunity cost of not protecting land. Although opportunity costs are only a partial measure of the full economic impact of conservation, they are also considered to be an appropriate measure for accounting for the social costs of conservation (Adams et al. 2010). This study made use of spatially explicit economic values for thirteen ecosystem services as calculated and mapped by Turpie et al. (submitted).

The holistic aim of the study was to assess how including spatial cost benefit data (e.g. Naidoo & Rickets 2006 and Chan et al 2011) in the cost surface would influence the spatial outcome of a national conservation assessment for South Africa (Figure 1). The conservation assessment was designed to identify priority areas for protected area expansion. In light of this overarching aim, the primary objectives of this study were to: (1) estimate the spatial variation in economic output of agricultural practices across South Africa, (2) identify conservation priority areas from reserve systems generated by different conservation assessment scenarios and (3), compare the identified priority areas in terms of area coverage, economic efficiency, and their ability to meet biodiversity targets. Specifically, this study addresses the following questions:

1. How does explicitly incorporating spatial cost benefit data into the cost surface influence the spatial distribution of priority areas? It is hypothesized that when cost benefit data are incorporated, priority areas will shift toward areas where conservation management makes more economic sense (economic benefits exceed or equal the economic costs).
2. What are the potential costs of failing to explicitly take ecosystem services into account? It is hypothesized that economic efficiency will decrease when the economic benefits of ecosystem services are not taken into account.
3. How does weighting economic data to account for the social costs of conservation influence the spatial distribution and overall cost of priority areas? It is hypothesized that the spatial distribution of the best reserve system will change and that the best reserve system will become more expensive.

These questions were investigated by way of four scenarios, each with different cost surface components. The biodiversity features and associated targets remained constant throughout the scenarios. The first three scenarios were designed such that each subsequent scenario included an additional cost surface component while the fourth scenario was a variation of the third. The cost surfaces of the respective scenarios included: (1) no economic data; (2) only economic costs (3), economic cost and benefit data and (4) economic cost and benefit data after accounting for the influence of conservation management on ecosystem service delivery. The first scenario acted as a baseline; the rationale being that priority areas would be identified based only on differences in biodiversity features because costs were essentially uniform across space. The rationale behind the second scenario was that it would reflect the influence of incorporating economic costs, as has been done by other studies when attempting to improve the cost-effectiveness of conservation planning. The third scenario built on the second by trading the economic benefits of ecosystem services off against the economic costs of conservation. The rationale was that spatial differences between priority sets from scenarios 2 and 3 would reflect the influence of including the economic benefits of

ecosystem services. Scenario 4 was different to scenario 3 in that it attempted to capture, in part, the influence of land management on ecosystem service. Generally speaking, protected land is likely to provide more in the way of regulating and cultural ecosystem services than unprotected land, and this thinking was reflected in scenario 4.

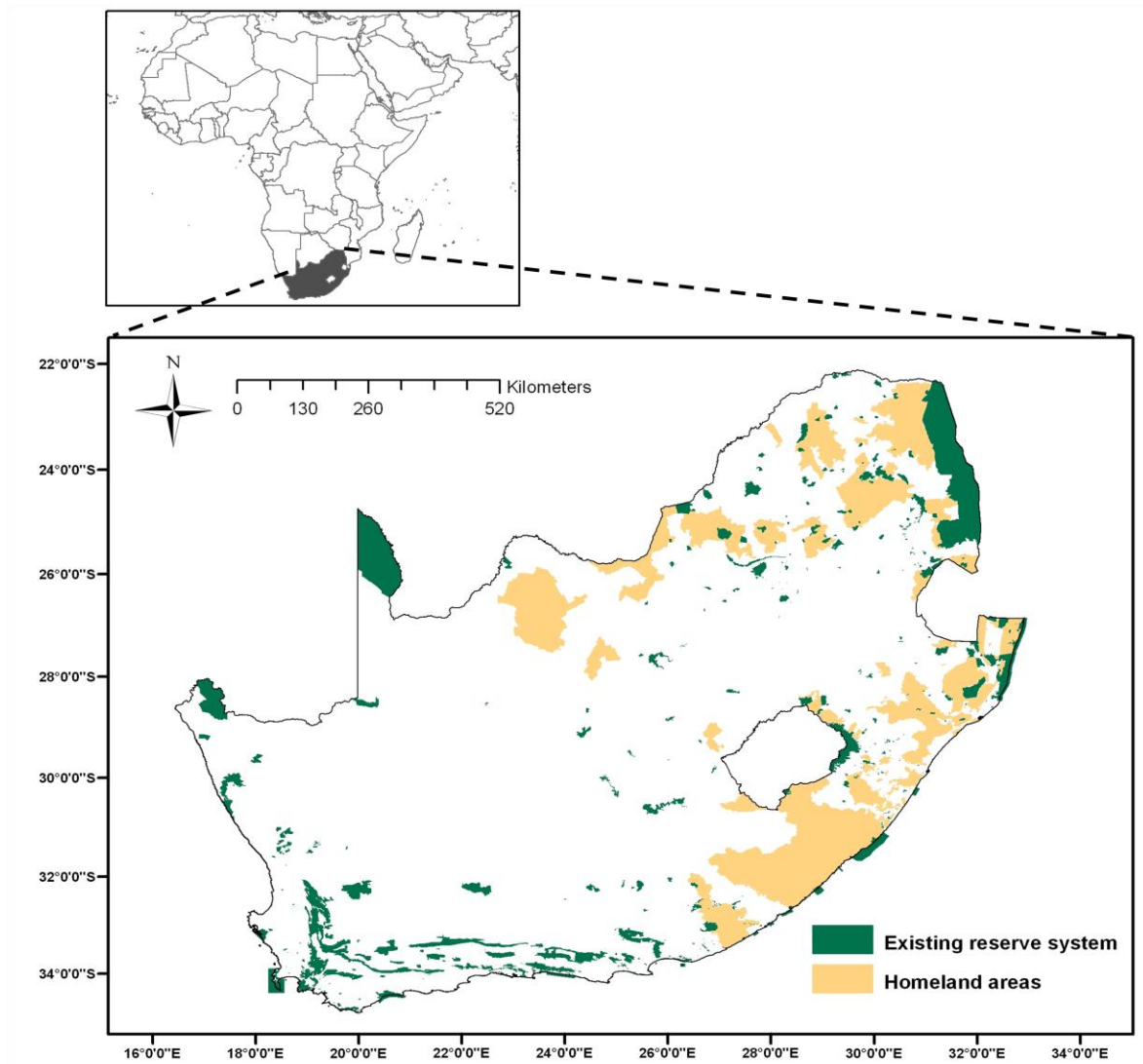


Figure 1: Map of the planning region, South Africa, displaying the existing protected area system (green) and the former homeland areas (yellow). The planning region was divided into 61,255 square planning units, most of which were 2025 ha. The existing protected area system was locked into all Marxan analyses

Methods

Planning region

The planning region was the whole of South Africa; which covers approximately 122 million hectares (Swaziland and Lesotho excluded), and extends 1 821 km NE-SW and 1 066 km SE-NW as a whole (Figure 1). South Africa is home to three biodiversity hotspots (Cape Floristic Region, Maputaland-Pondoland-Albany and Succulent Karoo) and supports a wealth of other ecosystems and species-rich landscapes (Wynberg 2002). In addition to being an important area for biological diversity, South Africa is a developing country that is home to some 51 million people (StatsSA 2012), many of which are poor and dependent on the land. For the most part, the country's development is intrinsically linked to the use of natural resources. The land-use situation in South Africa is further complicated by the fact that agricultural reform is seen as an important development strategy (Hall Researcher 2004). Agriculture is the predominant land-use, and although it contributes only a small percentage of the country's GDP, it is important in that it provides employment, increases food security, and earns foreign exchange (Goldblatt 2010). Commercial farming activities in South Africa include intensive crop production in winter rainfall and high summer rainfall areas, cattle ranching in the savannah areas, and small-stock farming in the Karoo (Goldblatt 2010) (Figure 2). Only 12% of the land surface is suitable for non-irrigated crop production, while 69% is suitable for grazing; making livestock farming the largest agricultural sector in the country (Conradie 2007; Goldblatt 2010). South Africa's existing protected areas cover *ca.* 6 % of the country, which falls short of the 12% required by the Convention of Biological Diversity (CBD) as of 2015 (Wynberg 2002; Snijders 2012). Furthermore, the existing protected areas do not adequately represent or protect the country's biodiversity because they were established based upon opportunism and are biased toward land with low production potential or high tourism value (Wynberg 2002; Reyers et al. 2007). Given this situation; new protected areas that protect and sustain biodiversity need to be established without significantly compromising agricultural production and development.

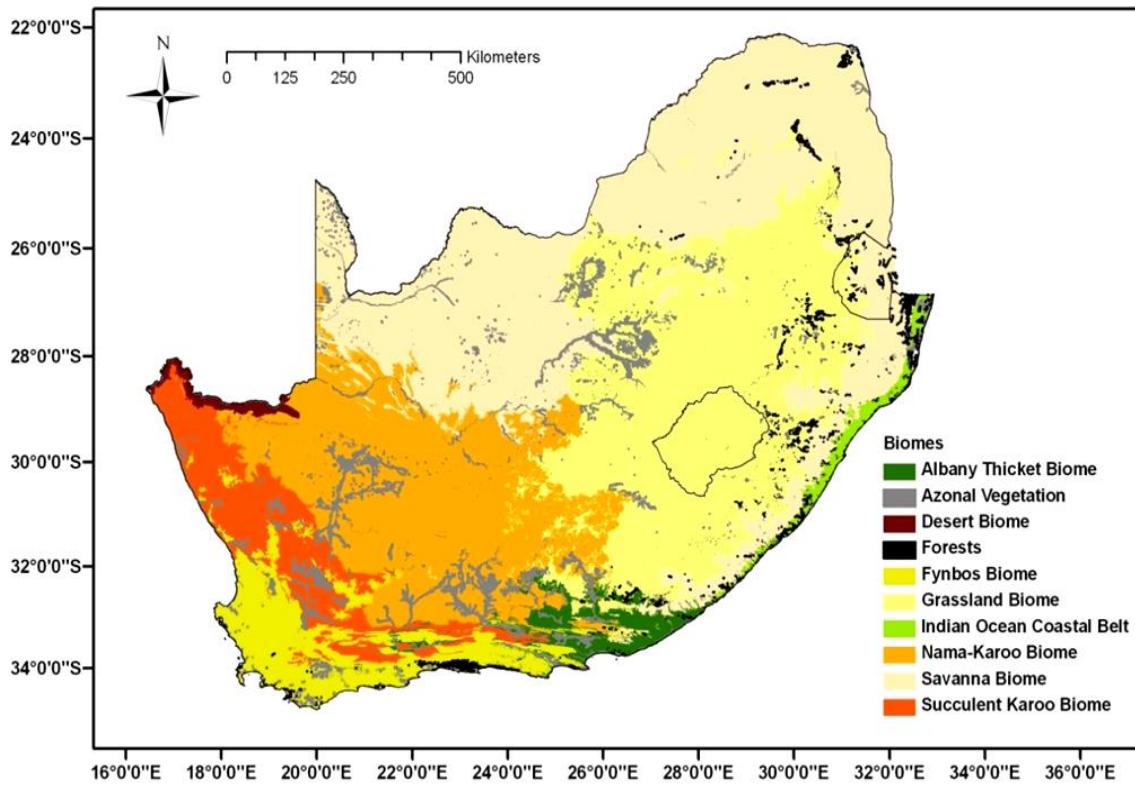


Figure 2: Biomes of South Africa, Lesotho and Swaziland (Rutherford et al. 2006)

Reserve design process

The site selection software Marxan (version 2.43) (Possingham et al. 2000) was used to identify priority areas for protected area expansion based upon conservation targets set for the vegetation bioregions of South Africa (Rutherford et al. 2006). Marxan is software that delivers decision *support* for reserve network design by identifying areas that meet all conservation targets when the objective function (cost) is minimized (Ball et al. 2009; Ardron et al. 2010). Marxan finds a range of “good” solutions using a heuristic algorithm known as simulated annealing (Ardron et al. 2010). The simulated annealing algorithm begins by generating a random solution from which new solutions are iteratively generated through sequential random changes (Beck & Odaya 2001; Ball et al. 2009). Each new solution is evaluated against the previous one to determine which solution meets the conservation goals for all the targets using the fewest number of sites (Beck & Odaya 2001). Marxan’s objective function includes a cost surface and two kinds of penalties and can be expressed simply as:

$$\text{Objective function} = \text{planning unit cost} + \text{boundary cost} + \text{species penalty factor}$$

Boundary cost is determined by the boundary length modifier (BLM), a weighting that can be used to control the spatial clumping of selected planning units, while the species penalty factor (SPF) assigns a penalty for failing to meet the specified conservation targets (Cook & Auster 2005; Chan et al.

2011). The reserve selection scenarios in this study differed from one another in that each scenario had a different cost surface. In all scenarios the SPF for each bioregion, with the exception of azonal and zonal & intrazonal forests, was proportional to the target and of a similar magnitude to the cost surface in *Scenario 1*. The SPF was set to ensure that all targets were satisfied by more than 99%. Guidelines provided by Ardron et al. (2010) were used to establish the parameters for the different planning scenarios. Where possible, the parameter values were maintained across scenarios. An adaptive annealing schedule was used and each run was finished with two-step iterative improvement. The results focussed on the “best” and “summed” solution outputs generated by Marxan.

Determining the appropriate planning unit size is somewhat subjective, but guidelines are provided by Ardron et al. (2010). Generally, smaller planning units will produce more efficient outcomes (Ardron et al. 2010). In previous Marxan-based conservation planning exercises, planning unit size has ranged from 100 ha units (Reyers et al. 2012) to variable unit sizes based upon subcatchments (Cameron et al. 2008). The conservation planning exercise in the National Protected Area Expansion Strategy (NPAES) for South Africa made use of hexagon planning units, most of which were 2000 ha (DEAT & SANBI 2009). Studies with objectives similar to those in this study have used 10 000 ha planning units (Carwardine et al. 2008, 2010). Considering the available data, the extent and objectives of this study; the study area was divided into 61 255 square planning units, most of which were 2025 ha.

Marxan was forced to include or exclude planning units depending on the protection status or extent of urbanization within the unit respectively. If > 20% of the planning unit was considered to be formally protected the planning unit was included or “locked in” (status = 2). Existing protected areas were locked into the solutions because the objective was to identify priority areas for protected area expansion. If >75% of the planning unit was considered to be informally protected the planning unit status was set to one to ensure that the unit was always selected in the initial random solution (status = 1). Where urbanisation was the predominant land-use (>25% of the unit) the relevant planning units were excluded or “locked out” (status = 3). Planning units where urban land-use covered less than 25% of the area were not excluded from selection because protected areas often border urban areas in South Africa (e.g. Cape Town).

Biodiversity targets were set for the 44 vegetation bioregions of South Africa which are classified according to the predominant vegetation types found within a region (Rutherford et al. 2006). Vegetation type has previously been used as a surrogate for species diversity in conservation planning exercises (Reyers et al. 2001). With the exception of the two forest bioregions, a target for each bioregion was set as outlined by the National Biodiversity Assessment (NBA) (Driver et al. 2012). The NBA outlines agreed-upon targets, which vary between 16% and 36% of the original extent of each terrestrial ecosystem type, that were calculated according to a species-area relationship

for each bioregion (Driver et al. 2012). Where the required data for this calculation were lacking the target was set to a generic 20% of the original extent (Driver et al. 2012). With the exception of the forests bioregions, Reyers et al. (2007) applied the same targets for vegetation types. The forest bioregions (azonal forests and zonal & intrazonal forests) had outlined targets of 100%. Marxan was not able to meet such high targets under any circumstance; therefore the targets were reduced to 70% and 80% for azonal forests and zonal & intrazonal forests respectively. Due to time constraints mammal, bird and invertebrate distribution data could not be obtained; hence the decision to conduct a course-scale conservation assessment based upon vegetation bioregions.

Reservation Scenarios: cost surfaces

Where information on cost is not available, reserve area can be used as a proxy for cost (based on the assumption that the larger the reserve size the more costly it will be to implement) or cost can be set as any other relevant social, economic or ecological measure (Ardron et al. 2010). Four conservation scenarios were examined in this study, each with a different cost surface. The features and associated method of inclusion are outlined in Table 1. The cost of each planning unit was represented by the average value (R) per hectare of land within each planning unit (Adams et al. 2010). A weighted average value for each planning unit was calculated based upon the relative proportions of the relevant administrative units (in the case of agricultural output) and grid cells (in the case of ecosystem service value) within each planning unit. This weighted value for agricultural output was then averaged according to the total area of each planning unit. In all scenarios, the cost of planning units considered to be under formal protection or urbanized was set to zero.

Table 1: Features included in all four scenarios, the method of inclusion (targeted, opportunity cost, economic) for each feature, and the proportion of current value. The proportion of current value represents the estimated difference in value between protected and unprotected land over time. The grazing ecosystem service is accounted for in the commercial and subsistence livestock production. The cost surface of *scenario 2* included only opportunity costs, *scenario 3* included opportunity costs and economic benefits, and *scenario 4* included opportunity costs and economic benefits after adjusting for the influence of land management on ecosystem service value.

Feature	Method of inclusion	Proportion of current value (%)
Vegetation Bioregions	Targeted	n/a
Commercial livestock production	Opportunity cost	n/a
Commercial cultivation	Opportunity cost	n/a
Subsistence livestock production	Opportunity cost	n/a
Subsistence cultivation	Opportunity cost	n/a
Natural resource utilization	Opportunity cost	n/a
Bio-prospecting	Economic benefit	80
Groundwater recharge	Economic benefit	25
Erosion prevention	Economic benefit	25
Water treatment	Economic benefit	25

Feature	Method of inclusion	Proportion of current value (%)
Pollination	Economic benefit	20
Pest control	Economic benefit	20
Disease control	Economic benefit	80
Flood attenuation	Economic benefit	25
Carbon storage	Economic benefit	15
Carbon sequestration potential	Economic benefit	80
Tourism value	Economic benefit	10

Scenario 1 (uniform cost): The cost surface was calculated based on the assumption that costs are uniform across space; therefore area was used as a proxy for the cost of each planning unit.

Scenario 2 (cost minimisation): The cost surface in this scenario included only the opportunity costs of foregone income from agriculture and natural resource utilization. Variations of this scenario were explored by weighting the value of subsistence uses of land and natural resources. Although such weightings have not been applied in conservation planning, they have been applied in the economic literature (Boadway 1976; Munda et al. 1994; Munda 1996, 2004). Given that the distributions of income are likely to vary across social groups, ways of integrating this distributional aspect into CBA need to be found (Munda 1996). Typically, distributional weights are introduced by weighting dissimilar social groups differently (Munda 1996). However, it is not clear how to derive such weights and who should attach them (Munda 1996); therefore a variety of weights were chosen to test the sensitivity of the scenario to different weightings. The weightings that are reported on in this study were 10, 100 and 1000.

Scenario 3 (net benefit maximization): The cost surface in this scenario incorporated a spatial CBA. The sum of economic benefits from regulating and cultural ecosystem services was subtracted from the sum of agricultural opportunity costs and economic benefits of provisioning ecosystem services for each planning unit. Carbon sequestration was excluded from the cost surface because the economic benefits of this ecosystem service would only be experienced post reserve establishment when degraded land is restored to its natural state. After running this scenario with values for the cost surface calculated as detailed above, it was evident that Marxan was selecting every planning unit with a negative cost value (irrespective of parameter values and SPF weightings). The result was reserve systems covering >60% of the country. To prevent over-selection, the entire cost surface was scaled to the lowest cost value and all cost values were made to be positive. The ratio between the highest and lowest cost values was used as a scaling increment.

$$x_1 = 1 ; x_2 = 1 + \left(\frac{\text{highest cost ppu}}{\text{lowest cost ppu}} \right) ; x_3 = x_2 + \left(\frac{\text{highest cost ppu}}{\text{lowest cost ppu}} \right) ; \dots$$

The preliminary outputs were checked to ensure that the solutions were still favouring planning units where the benefits exceeded the costs, as was the case prior to the scaling.

Scenario 4 (*maximizing the additional net benefits from conservation*): The cost surface in this scenario was calculated in the same manner as for scenario 3. The difference being that this scenario accounted for the influence of land management on ecosystem service delivery. Conservative estimates were made as to what the difference in ecosystem service value would be between protected and unprotected land over the next 20 years. The difference in value between protected and unprotected land was calculated as a percentage of the full economic values used in scenario 3, based upon the estimates given in Table 1. The estimates used in this study were based upon untested assumptions because there are no known relationships between land management and the value of ecosystem services. The estimates reflect the assumption that protected land will increase regulating and cultural service delivery. The estimate for tourism was low because it was assumed that tourists were relatively insensitive to changes in protected area coverage.

Marxan parameter sensitivity analyses

Sensitivity analyses were conducted to determine the most appropriate values for: the number of restarts, the number of iterations per simulation, and the BLM. The values were determined based upon the “best solution” generated for each scenario. Given time constraints, the number of restarts was determined by assessing how many restarts were required by Marxan to reach a point where the variation in spatial distribution between “best solutions” was minimal. Similarly, the number of iterations was determined by assessing how the efficiency of the “best solution” increased when the number of iterations was sequentially increased by an order of magnitude.

The appropriate BLM was determined by assessing the trade-off between reserve system cost and reserve system perimeter for each BLM value, while considering the spatial distribution of the resulting reserve system (Stewart & Possingham 2005). Impractical reserve systems are those that have very small clumps that are greatly dispersed, or have very large clumps with forced connections. The sensitivity analysis is required to identify the BLM value that produces a reserve system with the smallest perimeter for the lowest cost. The point just before the total cost of the “best” reserve system increases dramatically represents the ideal BLM (Stewart & Possingham 2005). This ‘ideal’ BLM was modified where necessary to obtain a practical reserve design. A BLM sensitivity analysis was conducted for each scenario individually. Due to time constraints, all scenarios were run with 100 restarts and 10^6 iterations. Arbitrary BLM values were chosen until the spatial distribution of the resulting reserve system was reasonable. Once an approximate BLM value had been established, a finer scale sensitivity analysis was conducted by sequentially increasing or decreasing the BLM value to obtain a BLM value that produced the most efficient reserve system in terms of the cost: perimeter trade-off.

Spatial distribution of agricultural value

Opportunity costs were defined as the foregone agricultural opportunities that would be encountered during protected area establishment. The variable was expressed as the average gross output per hectare ($\text{R}\cdot\text{ha}^{-1}$) generated by the current agricultural use of the land. Two broad groups of agricultural activities (crops and livestock) were considered based upon two different types of land tenure (private and communal). Agriculture under private land tenure was broadly classified as commercial agriculture, whereas that under communal tenure was taken to be either small-scale or subsistence agriculture, but the latter subcategories were not distinguished. Different data sources were used for commercial and communal-land agriculture. Agricultural opportunity costs were given by the sum of the value of commercial cultivation and livestock production, and subsistence crop and livestock production per planning unit (ppu). All spatial data were integrated in ArcGIS (version 9.31).

Commercial agriculture

Commercial agricultural data were extracted from the 2002 commercial agricultural census provincial reports (Appendix C). Although the most recent commercial agricultural census was conducted in 2007, detailed provincial-level data from this census are yet to be released. The 2002 economic data were updated to 2012 values, based upon the consumer price index, by an inflation factor of 1.38. An additional adjustment was made to account for the incomplete response rate. As of January 2004, only 87.4% of commercial farmers had completed and returned the 2002 census form. Income from cultivation and livestock production were treated separately. Cultivation refers to crop production and horticulture (fruit and vegetable production). Livestock production refers to animals sold and animal products (milk, cream, wool and mohair) sold. In both instances net income could not be calculated because the reported expenditure was not sufficiently detailed.

Gross income from cultivation was summarised at the district level. The average value ($\text{R}\cdot\text{ha}^{-1}$) of cultivation per district was calculated by dividing gross income (R) by total dry and irrigated area (ha) planted under crops and horticulture for each district. These values were applied to the spatial data on cultivated land (ha) within each district from the 2009 land-cover dataset. Finally, the values were summarised per planning unit, to produce an overall average value ($\text{R}\cdot\text{ha}^{-1}$) for cultivation ppu.

Gross income from livestock production was summarised at the district level. Gross income from livestock includes all income generated from game farming. The dataset was limited in that it provides no estimate of either the area grazed within each district or the numbers of livestock units (LSU) generating the livestock products. Thus, the gross output generated from livestock and livestock products was totalled for each district. These values were then mapped according to available grazing land and the associated grazing capacity of the land. The area of available grazing

land ppu was extracted from the 2009 land-cover dataset based on the assumption that all land classified as natural land was used for grazing, with the exception of natural land that fell within formal or informal reserves and within urban areas (Appendix C). Grazing capacity (ha.LSU^{-1}) was extracted from a dataset provided by the Department of Agriculture, Forestry & Fisheries (Appendix C). Grazing capacity was given as one of 14 grazing capacity categories. With the exception of transformed rangelands and the category given as >100 (where capacity was taken to be zero and 100 respectively), grazing capacity was taken to be the mean of the upper and lower limits of the category. These values were inverted to obtain a grazing capacity represented by LSU.ha^{-1} . The gross output per district was then mapped spatially within the district on the basis of grazing capacity. Finally, the values were summarised per planning unit, to produce an overall average value (R.ha^{-1}) for livestock production ppu.

Communal-land agriculture

Subsistence agriculture data came from the National Income Dynamics Study (NIDS) Wave1, 2008 (Appendix C); in which 7301 households were surveyed across the country. The study began in 2008, and the survey is repeated with the same households every two years. The purpose of the survey is to examine the livelihoods of individuals and households over time. The 2008 economic data were updated to 2012 values, based upon the consumer price index, by an inflation factor of 1.24. For consistency, values were also summarised in terms of gross output. Average gross income from subsistence agriculture per household (hh) (from hh that reported an income from subsistence agriculture) was calculated at the municipality level for crop and livestock farming separately. Gross income from mixed farming was allocated to either crop or livestock income based upon the ratio of income from crops to the income from livestock for the respective municipalities. Where the hh income was negative, gross income was set to zero; preventing an underestimation of the value of subsistence agriculture. Former homeland areas that were not represented in the NIDS dataset were allocated an average R.hh^{-1} for both crop and livestock production, based upon the district municipalities (with known income values) falling within the former homeland area. According to the NIDS dataset *ca.* 10 % of households surveyed generated an income from subsistence agriculture. The total number of hh's ppu was extracted from the 2001 Population Census dataset (Appendix C). The number of households was then multiplied by 0.1 to obtain an estimate of the number of hh's reliant on subsistence. The number of hh's reliant on subsistence agriculture was converted to household density (hh.ha^{-1}) ppu. The weighted average income per hh ppu for both crop and livestock production was multiplied by household density ppu, producing a value (R.ha^{-1}) for both categories of farming ppu. The value (R) of subsistence crop production per planning unit was obtained by multiplying the R.ha^{-1} value of crop production by the area of cultivated land (ha) within the respective planning units. Similarly, the value (R) of subsistence livestock production was calculated by multiplying the R.ha^{-1} value of livestock production by the area of grazing land within the respective planning units. Finally,

the values were summarised ppu, to produce an overall average value ($R\cdot ha^{-1}$) for subsistence agriculture ppu.

Spatial distribution in economic benefits from ecosystem services

The spatial distribution in economic benefits of ecosystem services across South Africa was as calculated and mapped by (Turpie et al. submitted). Thirteen different ecosystem services were included in this study: natural resource utilization, grazing, bio-prospecting, groundwater recharge (terrestrial vegetation), flood attenuation (wetlands), erosion prevention (natural vegetation), water treatment (wetlands), pollination (natural vegetation near crops), pest control (natural vegetation near crops), disease control (blackfly in rivers), carbon storage, carbon sequestration potential and tourism value. All 2008 values were updated to 2012 values by an inflation rate of 1.24. Grazing was accounted for in the commercial and subsistence livestock production and, like natural resource utilization, was considered an opportunity cost of conservation. The value of ecosystem services ppu was calculated as the sum of the weighted averages for all regulating and cultural services.

Identifying priority areas

Schmitt (2011) lists three ecological criteria for selecting priority areas: irreplaceability, vulnerability and representativeness. Irreplaceability is essentially a measure of conservation value, and can be defined as the likelihood of an area being required to meet the specified conservation targets (Cowling et al. 2003; Reyers et al. 2007). The “summed solutions” for each scenario were used to generate an irreplaceability score for each planning unit. The irreplaceability score was expressed as a proportion given by the number of times, out of 1000 runs, the planning unit was selected. A method for scheduling priority areas within a planning region is to plot selected variables on two axes (Margules & Pressey 2000). The first axis is the irreplaceability score. The second is vulnerability or threat; the risk of the area being transformed or losing its conservation value. Areas with high values for both variables should be considered high priority for conservation action (Margules & Pressey 2000). In this study the conservation statuses of vegetation types, as given by the National Spatial Biodiversity Assessment (2011) (Driver et al. 2012), were used as a proxy for vulnerability. The conservation status of a vegetation types is classified according to how much of the original extent remains. Vegetation types have been classified as least threatened ($\geq 80\%$ remains), vulnerable ($60\% < \text{remaining} < 80\%$), endangered ($< 60\%$ remains) and critically endangered (remaining extent \leq the conservation target of the vegetation type). For further details on the conservation status classification refer to Reyers et al. (2007). Least threatened, vulnerable, endangered and critically endangered conservation categories were allocated a value of one, two, three and four respectively. The conservation status of each planning unit was then calculated as a weighted average of conservation status values falling within the planning unit. A stepped approximation of a diagonal was used to delineate priority areas from non-priority areas for each scenario (Pressey & Taffs 2001).

Furthermore, all planning units with a selection frequency of 0.6 or more were included in an attempt to ensure that all conservation targets were in part included in the priority sets.

Analyses of reserve design solutions

Because the scenarios were run with different cost-surfaces in Marxan, *ad hoc* calculations were required to ensure fair comparisons in terms of efficiency (Chan et al. 2011). The outputs used to assess the efficiency of the respective scenarios were the best solution and priority set. All calculations excluded existing protected areas. Total area (ha), total opportunity cost (R), total management cost (R), total economic benefit before considering land management (R) and total economic benefit after considering land management (R) were calculated. Total opportunity cost included the value of provisioning services that would be lost if the land was conserved. Management costs were calculated according to the equation provided by Frazee et al. (2003):

$$\text{Annual management costs} \left(\frac{R}{ha} \right) = 66.55 (\text{area (ha)})^{0.6747}$$

Although this equation refers specifically to management costs of protected areas in the Cape Floristic Region, it is the best available for estimating the management costs of protected areas in South Africa. Management costs were updated from 2003 to 2012 values by a factor of 1.62 (based upon the consumer price index). Opportunity and management costs were summed and are referred to as “total” costs. These values were then used to calculate net value (total benefits – total costs) and three ratios: area: cost, area: benefit and cost: benefit.

Considering the objectives of this study, the most meaningful outputs from the scenarios were the selection frequencies and resulting priority sets. Therefore all further analyses were based upon these two sets of outputs. The spatial congruence of the solutions generated by the four scenarios was assessed using two methods. Firstly, the spatial congruence of priority sets (excluding existing protected areas) from pairs of scenarios were assessed using the kappa statistic (Richardson et al. 2006; Adams et al. 2010). The kappa statistic (κ) is an index that measures observed spatial agreement compared to that expected by chance (Adams et al. 2010). Kappa values range from -1 (complete disagreement) to 1 (complete agreement). For positive κ values; $\kappa < 0.2$ represents minimal agreement, $0.2 < \kappa < 0.4$ represents fair agreement, $0.4 < \kappa < 0.6$ represents moderate agreement, $0.6 < \kappa < 0.8$ represents good agreement, and $\kappa > 0.8$ represents very good agreement (Landis & Koch 1977). Second, the selection frequencies of planning units from pairs of scenarios were compared using Spearman rank correlation coefficient (ρ) (Adams et al. 2010). The Spearman rank correlation indicates the degree to which scenarios chose the same planning units in their solutions. A value of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation.

The ability of the priority sets (including current reserves) to meet the specified conservation targets was assessed by calculating how much (ha) of each bioregion was captured within each priority set. This value was expressed as a percentage of the respective target. The number of features whose targets were achieved and the number of features for which less than 20% of the target was achieved in each scenario were expressed as a percentage of the total number of features.

Results

Spatial distribution of opportunity costs

The estimated values of gross income ($\text{R}\cdot\text{ha}^{-1}$) generated by current commercial cultivation and livestock production were variable across the country (Figure 3). Where the estimated income from commercial cultivation was zero, the land was not capable of supporting cultivation, under formal protection, or fell within a former homeland where communal-land agriculture was the predominant land-use. Similarly, where no income from commercial livestock production was reported, the land was formally protected or fell within a former homeland. The spatial variation in gross income generated by commercial livestock production decreased in the Nama and Succulent Karoo biomes. This was due to decreased variability in the grazing capacity of the land and decreased resolution in the economic data. Values ($\text{R}\cdot\text{ha}^{-1}$) ppu for cultivation and livestock production were of the same magnitude. In sharp contrast to income generated from commercial agriculture, gross income ($\text{R}\cdot\text{ha}^{-1}$) from subsistence (communal-land) agriculture was substantially lower (Figure 4). The combined value of subsistence crop and livestock production was consistently two orders of magnitude lower than both commercial cultivation and commercial livestock production. Spatial variation in opportunity costs associated with conservation across South Africa is given in Figure 5. Total opportunity cost ppu ranges from negligible to *ca.* 144 300 $\text{R}\cdot\text{ha}^{-1}$. Opportunity costs were higher in the southern and north eastern parts of the country. The inclusion of natural resource utilization increased the opportunity costs of conservation in the former homelands. Irrespective of its production potential, land that is currently protected had no associated opportunity cost because agricultural production and resource utilization is prohibited in these areas.

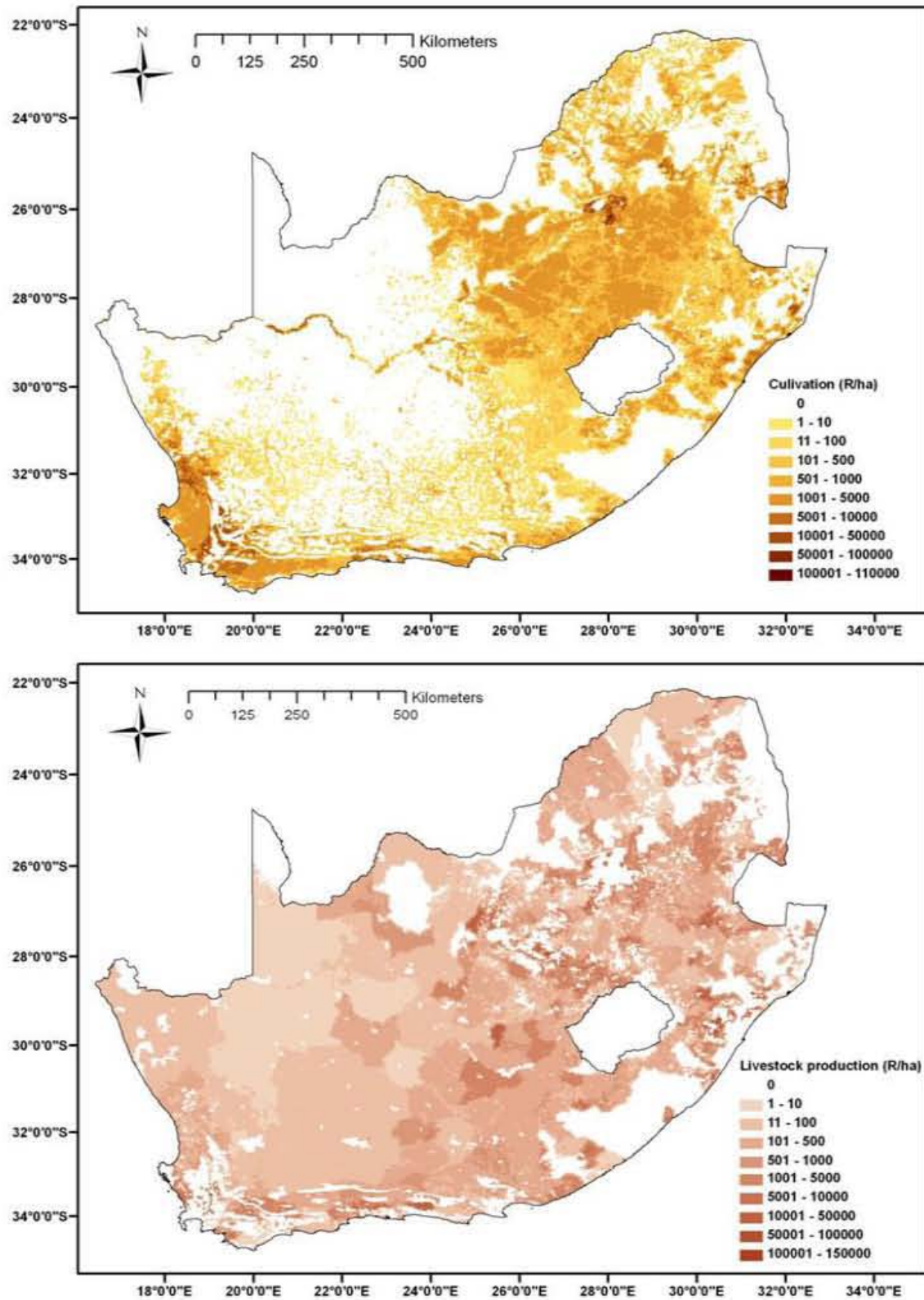


Figure 3: Maps of the spatial distribution in average gross income (R.ha⁻¹) per planning unit from current commercial cultivation (top) and commercial livestock production (bottom) across South Africa.

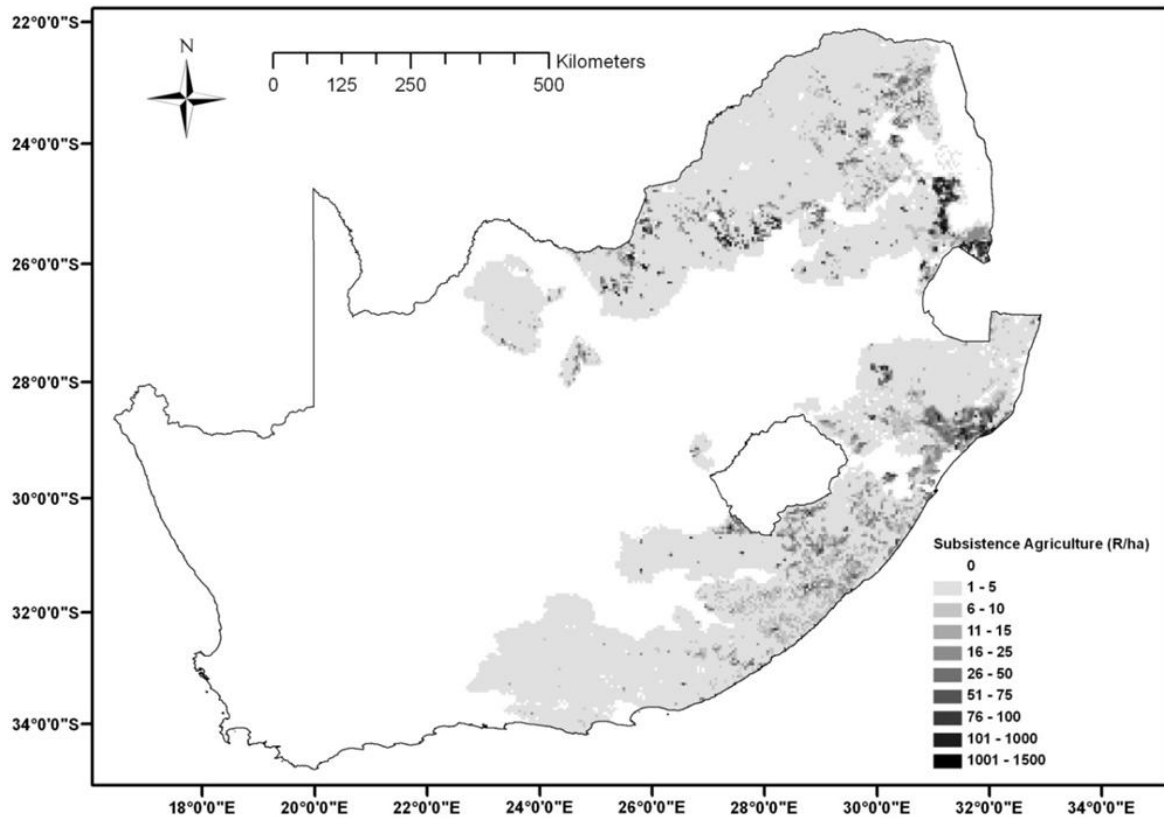


Figure 4: Map of the spatial distribution in average gross income from communal-land (subsistence) agriculture (R.ha⁻¹) per planning unit across the former homelands of South Africa. The values represent the average gross income from both crop and livestock production.

Spatial distribution of economic benefits from ecosystem services

The spatial distribution of economic benefits generated by regulating and cultural ecosystem services (Figure 6) was similar to that of opportunity costs (Figure 5). Economic benefits were higher in the southern, eastern and north eastern parts of South Africa. Despite having a similar spatial distribution, economic benefits seem less spatially variable than opportunity costs; particularly at a regional or local level. As with economic output from commercial livestock production there was little variation in the value of ecosystem service delivery in the Nama and Succulent Karoo biomes. Economic benefits of ecosystem service delivery ranged from *ca.* 25 to 25 200 R.ha⁻¹ ppu. Figure 6 represents the value of ecosystem services before land management was taken into consideration. The spatial CBA of opportunity costs and economic benefits of ecosystem services is displayed in Figure 7. Where the economic benefits of ecosystem services exceeded the opportunity costs, conservation management would make economic sense. When the influence of land management was taken into consideration, the total area suitable for conservation (from an economic efficiency perspective) remained in the same regions (western interior, eastern coastline and the far north) but contracted substantially. The opportunity costs of conservation typically outweigh the economic benefits of

conservation in areas where commercial cultivation or mixed farming are the predominant land-use types (southern and eastern interior parts of the country).

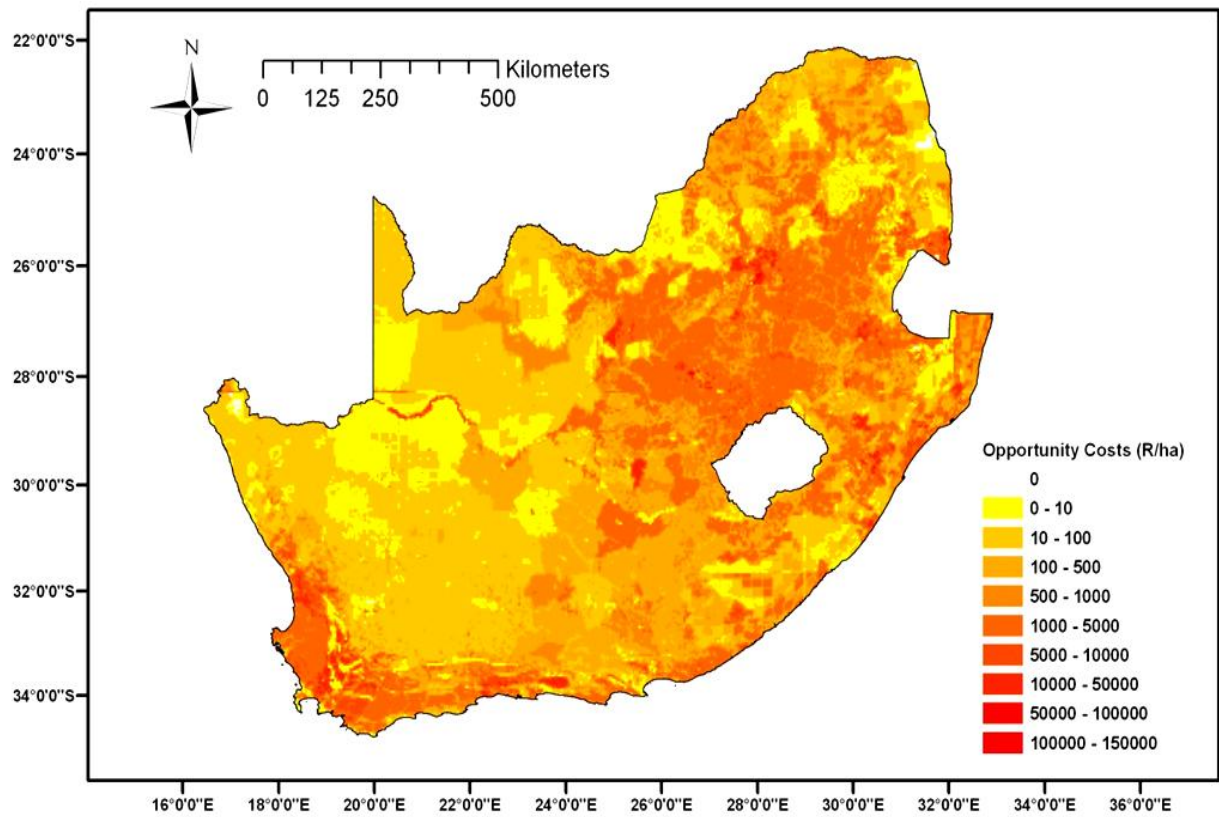


Figure 5: Map of the spatial distribution in opportunity costs, given as the sum of agricultural production (commercial and subsistence) and natural resource utilization, given as average value ($\text{R}\cdot\text{ha}^{-1}$) per planning unit. This map is also a graphical representation of the cost surface applied in scenario 2.

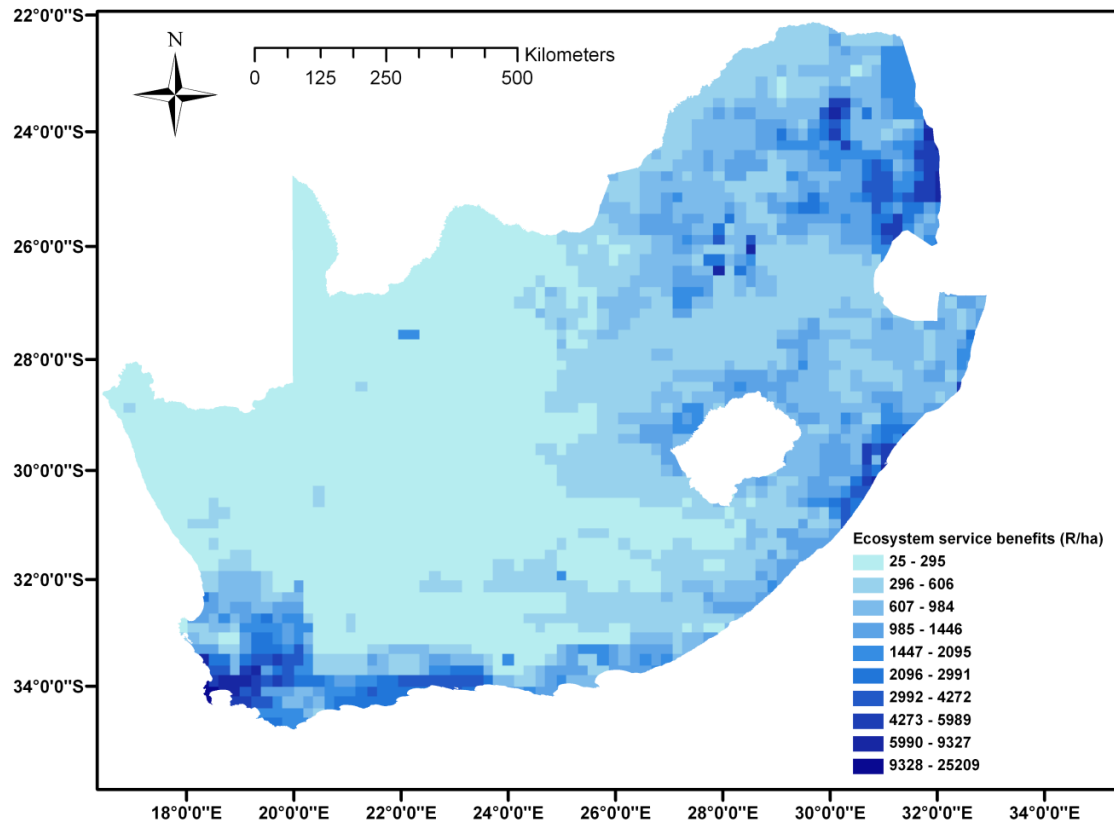


Figure 6: Map of the spatial distribution in average value ($\text{R}\cdot\text{ha}^{-1}$) per planning unit generated by regulating and cultural ecosystem service delivery (Turpie et al. submitted). The values displayed here represent the value of ecosystem service delivery before the influence of land management was taken into consideration.

Marxan parameter sensitivity analyses

With the exception of the BLM parameter, all parameter values were maintained across the scenarios (Table 2). Given the time constraints, 1000 restarts was sufficient for Marxan to account for the variation in reserve designs, while 10^7 iterations allowed Marxan to identify the most efficient reserve networks. As expected, Marxan was sensitive to changes in BLM value for all four scenarios. BLM values ranged across four orders of magnitude, with scenario 1 having the highest value and scenario 4 the lowest value. Despite these differences, all of the chosen BLM values were situated at similar positions along their respective cost: perimeter trade-off curves (Figure 8) and produced spatially cohesive reserve systems (Appendix A).

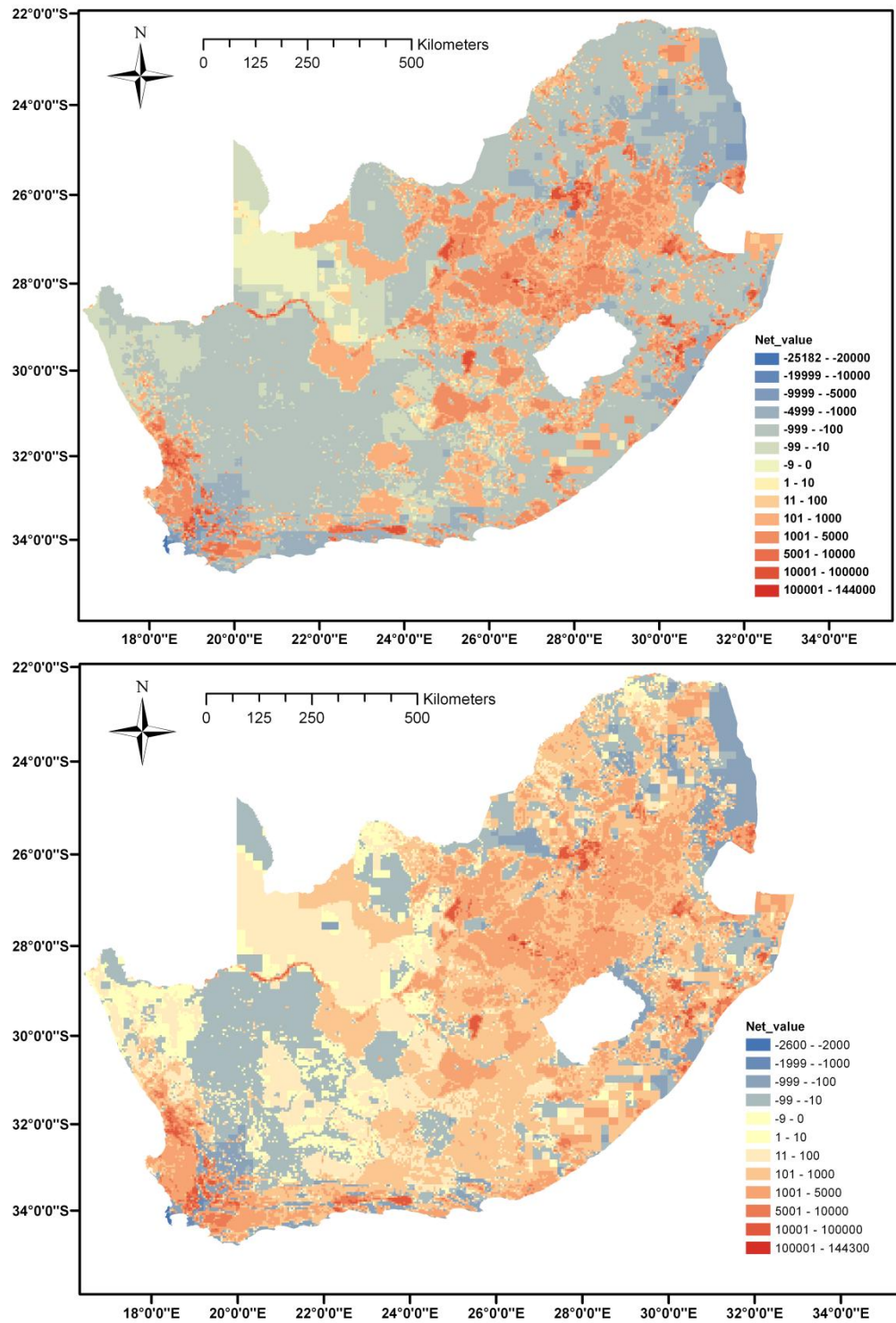


Figure 7: Spatial cost benefit analysis representing the trade-off between the opportunity costs of foregone agricultural production and natural resource utilization, and the economic benefits of ecosystem service delivery. The top map is the result of the cost benefit analysis prior to considering the influence of land management. The bottom map is the result of the cost benefit post considering the influence of land management. Red represents the areas where the opportunity costs exceed the economic benefits; blue represents the areas where the economic benefits exceed the opportunity costs. Beige represents the areas where the difference between the economic benefits and opportunity costs was negligible.

Table 2: Marxan parameter values for all four scenarios, as determined by sensitivity analyses where appropriate. Starting proportion refers to the proportion of planning units that were considered in the initial run. Species missing refers to the proportion of the conservation target that determines whether a target is considered to be met or not.

Parameter	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Restarts	1000	1000	1000	1000
Boundary Length Modifier	100	0.2	10	0.075
Iterations	10^7	10^7	10^7	10^7
Starting proportion	1	1	1	1
Species missing	0.99	0.99	0.99	0.99

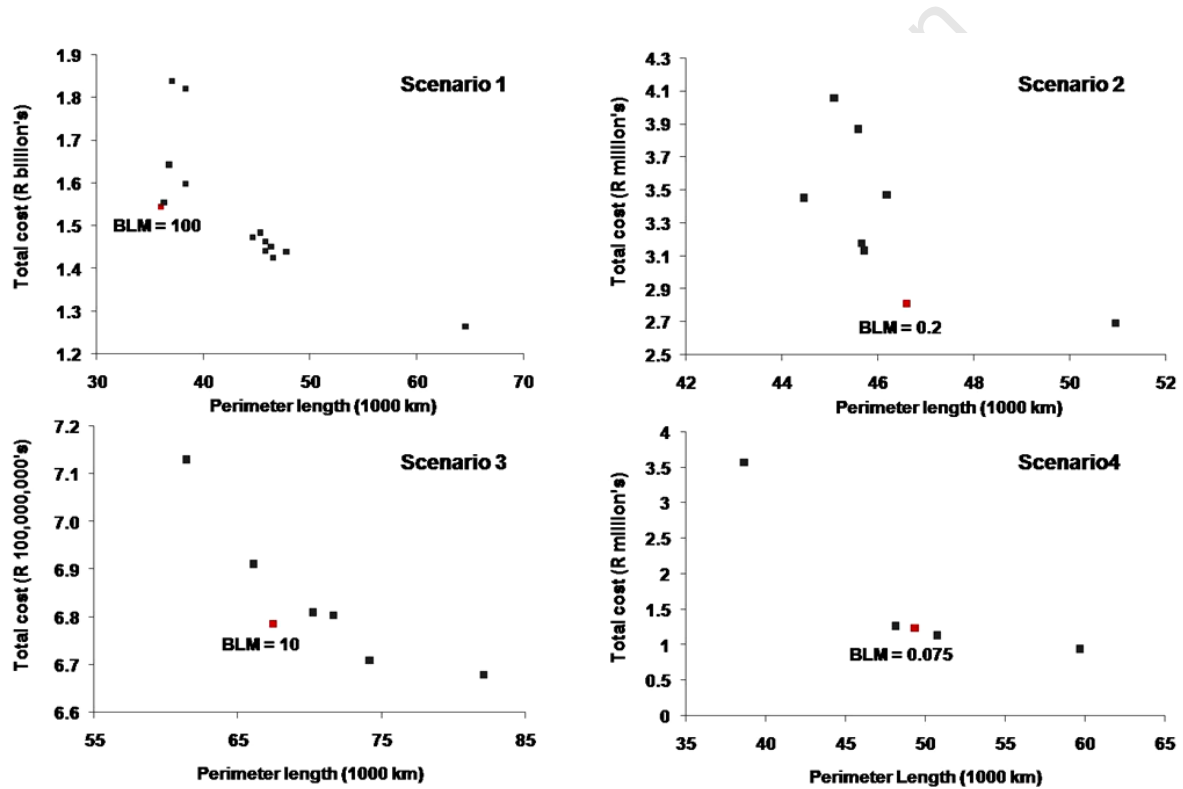


Figure 8: The results of the trade-off between total reserve network cost (R) and reserve network perimeter (km) for the boundary length modifier (BLM) sensitivity analysis for each scenario, based upon the best solution from 100 restarts with 106 iterations. The red point indicates the trade-off produced by the most appropriate BLM value, which is indicated in text below the point. Scenario 1, 2, 3 and 4 are arranged clockwise from top left. Scenario 1: BLM = 100, Scenario 2: BLM = 0.2, Scenario 3: BLM = 10, Scenario 4: BLM = 0.075.

Reserve Selection Scenarios

Best solutions

While, the “best” solutions generated by Marxan for each scenario (Appendix A) were not the primary focus of this study, a comparison of *ad hoc* calculations pertaining to these solutions indicated that economic efficiency improved when the economic costs and benefits of ecosystem services were included (Table 3). Prior to considering the influence of land management on the value of ecosystem services, scenario 1 (*uniform cost*) was the only scenario in which the net value of the reserve system was negative. The net value of scenario 3 (*net benefit maximization*) and scenario 4 (*additional net benefit maximization*) was substantially higher than that of scenario 2 (*cost minimization*). Although scenario 4 produced a reserve system that conserved substantially more land than scenario 1, the total cost of scenario 4 was a fifth of the total cost of scenario 1. According to the area: cost ratio, scenarios 4 and 2 performed the best in terms of minimizing costs per unit area. The area: benefit ratio indicated that when economic benefits were explicitly incorporated (scenarios 3 and 4) the reserve systems were able to maximize the benefits per unit area. Scenario 4 performed best in terms of the cost: benefit ratio. By contrast, scenario 1 performed poorly, with a cost: benefit ratio more than four times greater than scenarios 2 and 3 and almost nine times greater than scenario 4. When the influence of land management was considered, the net value of all four reserve systems was negative. However, scenario 4 was still the most efficient in terms of maximizing the benefits per unit cost, and had the highest net value. Irrespective of whether land management was considered, according to the cost: benefit ratio scenario 2 was more efficient than scenario 3 but less efficient than scenario 4.

Selection frequencies

All restarts in each scenario produced feasible results (all specified conservation targets were met without encountering penalty or generating a shortfall). The selection frequencies for each planning unit per scenario are displayed in Figure 9. The areas immediately surrounding the existing reserve system had the highest selection frequencies in scenario 1. The exception to this pattern was the Wild Coast along the eastern coastline, where high selection frequencies occurred despite the absence of existing terrestrial reserves. It was by chance that this area of high selection frequency bordered a marine protected area(s). While all of the above was true to some extent for scenarios 2, 3 and 4, the selection frequencies of the remaining three scenarios appeared to be less dependent on (or related to) the presence of existing protected areas. In scenario 2 and scenario 3, the arid western, central and eastern interior regions have higher selection frequencies than in scenario 1. The same is true of scenario 4 when compared to scenario 1, however, scenario 4 is characterised by a large area of high selection frequencies in the western interior.

Table 3: Comparison of the 'best' solution, excluding existing reserves, for each scenario in terms of total area (ha), opportunity costs (R), management costs (R), total costs (R), ecosystem service benefits (R), net value (R), and three ratios: area: cost ratio, area: benefits ratio and cost: benefit ratio. The benefits (R), area: benefit ratio and cost: benefit ratio were calculated when the influence of conservation (land) management on the economic benefits of ecosystem services was not considered (excluded) and when it was considered (included). Net value was calculated as the benefits – total costs. M = millions, B = billions.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Total area (ha)	23.05 M	22.65 M	22.05 M	32.98 M
Opportunity costs (R)	24.70 B	4.15 B	8.30 B	4.44 B
Management costs (R)	0.21	0.21	0.20	0.30
Total costs	24.91	4.35	8.50	4.73
Area: cost ratio	0.0009	0.0052	0.0026	0.0070
<i>Land management excluded</i>				
Total benefits (R)	14.60 B	11.54 B	19.77 B	22.79 B
Net value (R)	-10.31 B	7.18 B	11.27 B	18.05 B
Area: benefit ratio	0.0016	0.0020	0.0011	0.0014
Cost: benefit ratio	1.71	0.38	0.43	0.21
<i>Land management included</i>				
Total benefits (R)	2.34 B	1.85 B	2.91 B	3.52 B
Net value (R)	-22.57 B	-2.5 B	-5.59 B	-1.22 B
Area: benefit ratio	0.0098	0.0122	0.0076	0.0094
Cost: benefit ratio	10.63	2.35	2.92	1.35

Spearman's rank correlation (ρ) was used to evaluate the degree to which scenarios selected the same planning units (Table 4). When the existing reserves were included, the correlations between pairs of scenarios were moderate to high, ranging from 0.44 to 0.91. Excluding existing reserves, the correlations (ρ) decreased fractionally, ranging between 0.4 and 0.9. Of the four scenarios, the spatial distribution of scenario 1 was most similar to that of scenario 3 ($\rho = 0.64$), while scenario 2 was most similar to scenario 4 ($\rho = 0.90$). Scenario 3 was however also quite similar to both scenario 2 ($\rho = 0.72$) and scenario 4 ($\rho = 0.71$).

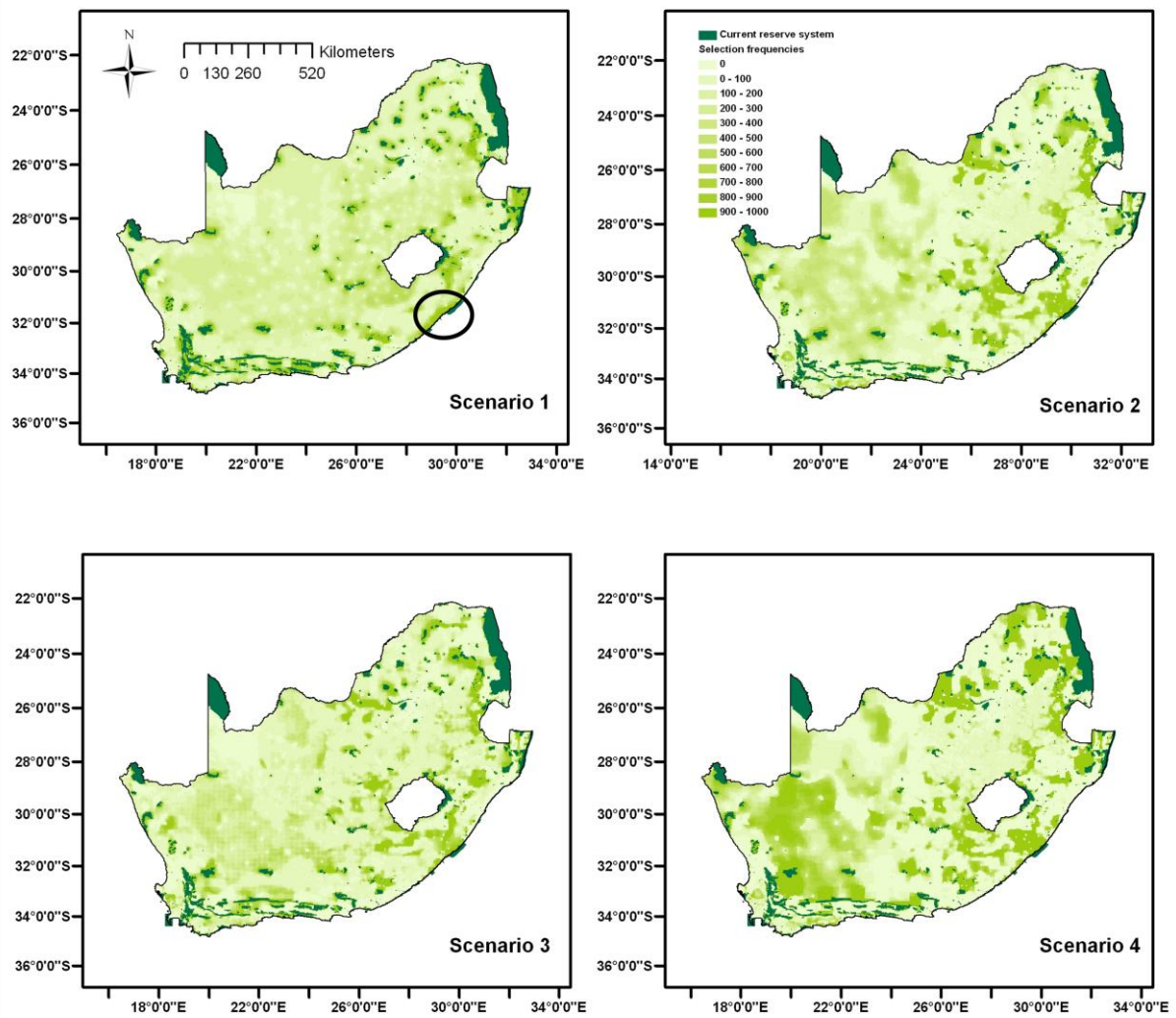


Figure 9: Maps of the spatial distribution in selection frequency (of 1000 restarts) for all four scenarios. The legend, scale bar and north arrow applies to all four maps. The current reserve system was locked into the scenarios (selected frequency = 1000), but has been indicated in dark green for the purposes of presentation and interpretation. The black ring in the top map indicates the location of the Wild Coast, as referred to in text.

Table 4: Spearman rank correlations (ρ) for correlations between the selection frequencies of pairs of scenarios. Values above the diagonal exclude areas locked into the reserve networks; values below the diagonal include areas locked into the reserve networks. All values were significant ($p < 0.01$)

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Scenario 1	1	0.49	0.64	0.40
Scenario 2	0.54	1	0.72	0.90
Scenario 3	0.66	0.75	1	0.71
Scenario 4	0.44	0.91	0.74	1

Priority areas

As with the best solutions (Table 3), a comparison of the *ad hoc* calculations pertaining to the priority sets indicated that the efficiency results were as expected (Table 5). The priority set identified from Scenario 1 was consistently the most inefficient; furthermore, it covered the least amount of land and yet was the most expensive. Prior to considering the influence of land management on the economic benefits of ecosystem services, scenario 1 was the only scenario in which the priority set generated a net loss. Scenario 3 was able to maximise the economic benefits per unit area the best, irrespective of whether or not land management was considered. In terms of economic efficiency, scenario 4 was able to maximize the economic benefits per unit cost the best. Prior to considering land management total costs were 17%, 32%, 36% and 164% of the economic benefits in scenario 4, scenario 3, scenario 2 and scenario 1 respectively. When the influence of land management was incorporated, all four priority sets generated a net loss. However, when economic costs and benefits were considered the net loss decreased substantially. Scenario 4 was still able to maximize the economic benefits per unit cost the best. When the influence of land management was considered, scenario 2 and scenario 3 were very similar in terms of maximizing the benefits per unit cost (2.3 vs. 2.28 respectively). In terms of economic efficiency, the NPAES focus areas were not as inefficient as the priority set of scenario 1, but were less efficient than the priority sets from scenarios 2, 3 and 4. The difference in economic efficiency between the priority sets from the different scenarios may be dampened by the priority area identification method used in the study. Planning units that were considered to be highly vulnerable were on average more expensive than all other planning units (Appendix B). However, highly vulnerable planning units also generated on average more economic benefits than all other planning units.

Table 5: Comparison of the priority sets, excluding existing reserves, for each scenario and the NPAES focus areas in terms of total area (ha), opportunity costs (R), management costs (R), total costs (R), ecosystem service benefits (R), net value (R), and three ratios: area: cost ratio, area: benefits ratio and cost: benefit ratio. The benefits (R), area: benefit ratio and cost: benefit ratio were calculated when the influence of conservation (land) management on the economic benefits of ecosystem services was not considered (excluded) and when it was considered (included). Net value was calculated as the benefits – costs. M = millions, B = billions.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4	NPAES Focus Areas
Total area (ha)	9.93M	14.29M	13.05M	29.48M	11.71M
Opportunity costs (R)	14.00B	3.10B	5.12B	3.63B	6.65B
Management costs (R)	0.09B	0.13B	0.12B	0.27B	0.12B
Total costs (R)	14.09B	3.23B	5.23B	3.89B	6.78B
Area: cost ratio	0.0007	0.0044	0.0025	0.0076	0.0017
<i>Land management excluded</i>					
Benefits (R)	8.55B	9.01B	16.25B	22.16B	7.25B
Net value (R)	-5.54B	5.78B	11.01B	18.27B	0.48B
Area: benefit ratio	0.0012	0.0016	0.0008	0.0013	0.0016
Cost: benefit ratio	1.65	0.36	0.32	0.18	0.93
<i>Land management included</i>					
Benefits (R)	1.35B	1.41B	2.30B	3.39B	1.17B
Net value (R)	-12.74B	-1.82B	-2.94B	-0.50B	-5.61B
Area: benefit ratio	0.0074	0.0102	0.0057	0.0087	0.0100
Cost: benefit ratio	10.44	2.30	2.28	1.15	5.79

Excluding the current reserve system, the priority set from scenario 1 comprised 5052 planning units; less than the priority sets from scenario 2 (7205 planning units) and scenario 3 (6514 planning units). The priority network identified from scenario 4 was substantially larger, comprising 14 766 planning units. The spatial distributions of the priority sets are displayed in Figure 10. With the exception of the eastern coastline the priority areas identified from scenario 1 were primarily situated adjacent to existing protected areas. The same cannot be said for the remaining three scenarios. Priority areas identified from scenario 2 typically covered large tracts of land in the eastern regions of the country; while priority areas in the western regions typically surrounded existing protected areas. The priority areas of scenario 3 were similar to those of scenario 2 in terms of spatial distribution, but differed in that they generally covered less land. The priority set from scenario 4 covered substantially more land than any other priority network (Table 5), and differed from the first three scenarios in that

a very large priority area was located in the arid west of the country. Of the four scenarios, the priority set from scenario 4 was most similar to the NPAES focus areas in terms of spatial distribution.

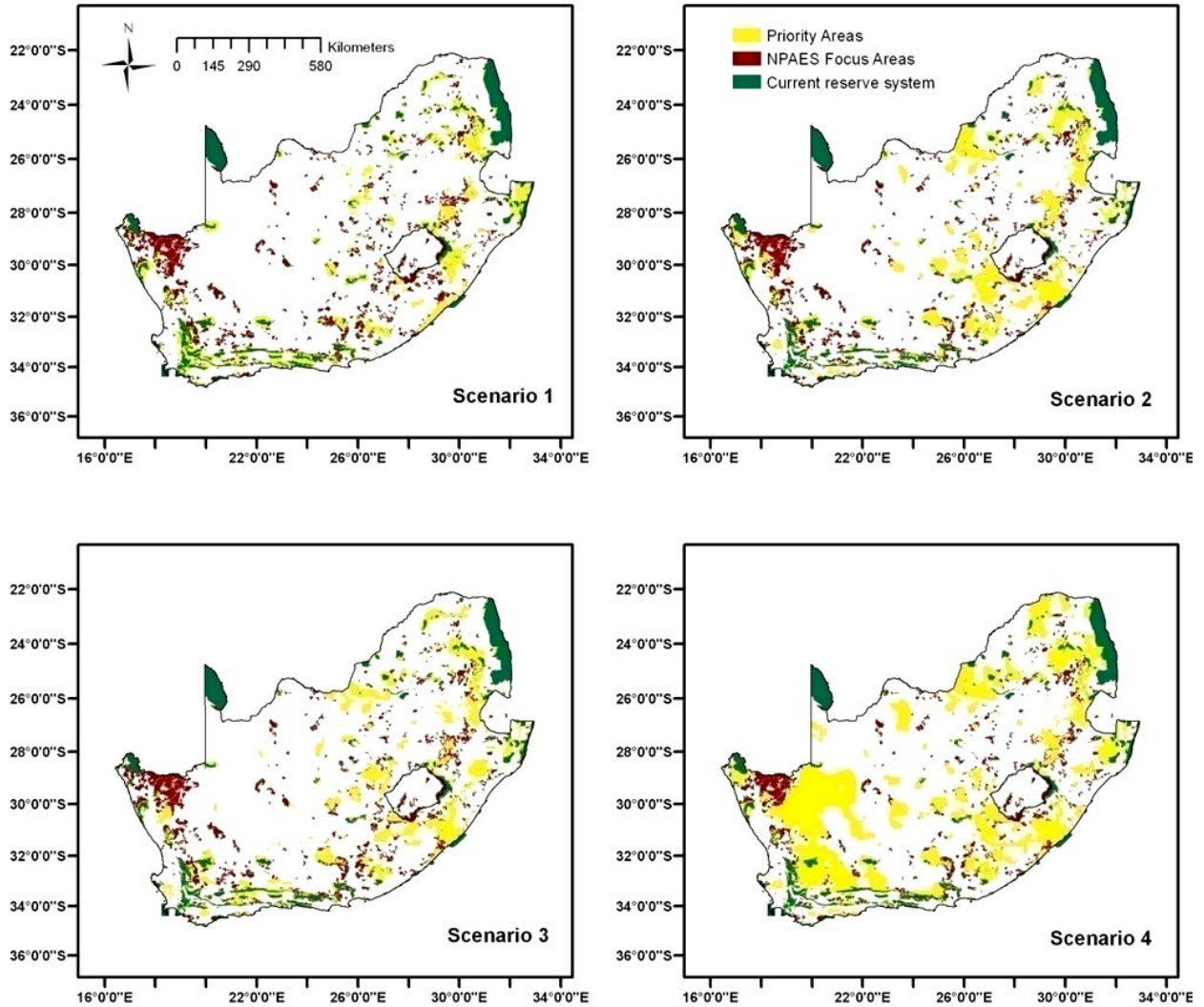


Figure 10: The spatial distribution on the priority sets (yellow) identified for the four scenarios. The existing reserve system is excluded from these networks but represented in green. The focus areas identified during the National Protected Area Expansion Strategy are represented in red. The north arrow, scale bar and legend apply to all four maps.

The strength of spatial agreement between the different priority sets (excluding existing reserves) was minimal to moderate according to the kappa statistics (Table 6). The priority set from scenario 1 was fairly similar to those from scenario 2 and scenario 3 ($0.2 < \kappa < 0.4$). There was however little agreement between the priority sets from scenario 1 and scenario 4 ($\kappa < 0.2$). Spatial

agreement was greatest between the priority set from scenario 2 and those from scenarios 3 and 4 ($\kappa = 0.54$ in both instances). There was fair agreement between the priority sets from scenarios 3 and 4.

Table 6: A comparison of the kappa statistics (κ) for the spatial agreement between the priority networks from pairs of scenarios. The existing reserves were excluded from the priority areas in this analysis. All values were significant with $p < 0.001$; Standard error is indicated in brackets.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Scenario 1	1			
Scenario 2	0.33 (0.0073)	1		
Scenario 3	0.34 (0.0074)	0.54 (0.0059)	1	
Scenario 4	0.13 (0.0061)	0.54 (0.0046)	0.37 (0.0053)	1

The ability of the priority areas to meet conservation targets was assessed and is given in Table 7. Scenario 4 achieved the targets for 29 (67%) vegetation bioregions while scenario 1, scenario 2 and scenario 3 achieved the targets for 18 (42%), 19 (44%) and 16 (37%) vegetation bioregions respectively. There was little variation between scenarios as to which bioregion targets were met. Together, the scenarios achieved $< 20\%$ of the target for ten vegetation bioregions. Of these ten bioregions, scenario 1 and scenario 2 achieved $< 20\%$ of the target for six of them, and scenario 3 achieved $< 20\%$ of the target for seven of them. Scenario 4 achieved $> 20\%$ of the target for all 43 vegetation bioregions; with the lowest percentage of a target achieved at 64%.

Table 7: A comparison of the priority sets (including existing reserves) in terms of their ability to meet the specified targets for the vegetation bioregions of South Africa. The percentage of targets met in each scenario were compared. 'Targets exceeded' refers to the number of targets that were exceeded ($>100\%$). Instances where less than 20% but more than 10% of the target was met are highlighted in pink. Instances where less than 10% of the target was met are highlighted in yellow. The target for each bioregion was expressed in millions of hectares (M.ha).

		Scenario 1	Scenario 2	Scenario 3	Scenario 4
Targets exceeded (%)		41.9	44.2	37.2	67.44
Less than 20% of target met (%)		14.0	14.0	16.3	0.00
Bioregion	Target (M. ha)	% target met	% target met	% target met	% target met
Knersvlakte	1417	Exceeded	98.14	92.22	Exceeded
Seashore Vegetation	91	Exceeded	Exceeded	Exceeded	Exceeded
Karoo Renosterveld	1501	31.22	21.47	27.84	Exceeded
Northwest Fynbos	4274	Exceeded	Exceeded	Exceeded	Exceeded

		Scenario 1	Scenario 2	Scenario 3	Scenario 4
West Strandveld	562	65.27	Exceeded	92.96	Exceeded
Namaqualand Sandveld	2365	73.00	95.75	63.05	Exceeded
Rainshadow Valley Karoo	4463	Exceeded	91.22	Exceeded	Exceeded
Estuarine Vegetation	39	Exceeded	Exceeded	Exceeded	Exceeded
Trans-Escarpment Succulent Karoo	2775	24.62	18.89	36.64	Exceeded
Freshwater Wetlands	318	Exceeded	Exceeded	Exceeded	Exceeded
Alluvial Vegetation	2733	61.06	89.31	70.61	89.66
West Coast Renosterveld	1582	36.32	28.35	92.41	Exceeded
Inland Saline Vegetation	4285	38.95	56.97	10.59	Exceeded
Southwest Fynbos	3335	36.02	45.89	57.87	Exceeded
Western Fynbos-Renosterveld	2790	61.69	70.08	52.41	Exceeded
East Coast Renosterveld	2307	Exceeded	Exceeded	Exceeded	97.23
Southern Fynbos	912	Exceeded	Exceeded	Exceeded	Exceeded
Zonal & Intrazonal Forests	3591	47.92	57.29	Exceeded	98.48
South Strandveld	304	Exceeded	Exceeded	Exceeded	90.07
South Coast Fynbos	974	Exceeded	Exceeded	Exceeded	88.94
Upper Karoo	25312	1.19	1.09	14.05	74.90
Dry Highveld Grassland	30641	1.75	2.77	0.90	90.67
Lower Karoo Bioregion	5098	79.44	Exceeded	12.55	Exceeded
Albany Thicket	5534	Exceeded	Exceeded	80.15	Exceeded
Eastern Fynbos-Renosterveld	3949	34.43	32.99	Exceeded	Exceeded
Eastern Strandveld	94	Exceeded	Exceeded	Exceeded	98.93
Southern Namib Desert	241	Exceeded	Exceeded	Exceeded	Exceeded
Bushmanland Bioregion	20135	0.54	0.46	33.11	Exceeded
Richtersveld Bioregion	1163	25.96	22.16	7.81	Exceeded
Namaqualand Cape Shrublands	214	Exceeded	Exceeded	95.78	98.46
Gariep Desert	2143	87.94	Exceeded	Exceeded	Exceeded
Namaqualand Hardeveld	5419	2.78	3.57	31.48	85.16
Eastern Kalahari Bushveld	19882	8.67	8.87	1.05	64.39
Kalahari Duneveld	6821	44.75	56.51	23.22	Exceeded
Sub-Escarpment Savanna	8881	47.17	66.06	42.69	Exceeded
Sub-Escarpment Grassland	17261	47.50	47.46	44.68	Exceeded
Drakensberg Grassland	11321	38.35	69.23	72.37	95.73
Indian Ocean Coastal Belt	3570	Exceeded	Exceeded	Exceeded	Exceeded
Mesic Highveld Grassland	29791	23.01	36.43	49.07	92.42
Azonal Forests	161	Exceeded	Exceeded	Exceeded	99.98
Lowveld	12556	Exceeded	Exceeded	32.16	Exceeded
Central Bushveld	28410	0.56	0.57	87.12	Exceeded
Mopane	4877	Exceeded	Exceeded	3.28	Exceeded

Weighted opportunity cost scenarios

Weighting the subsistence value of land and natural resource utilization only influenced the spatial distribution of the “best solution” when a weighting factor > 100 was applied (Figure 11). The impact of weighting the data was felt primarily in the former homelands, with little displacement of protected areas into other parts of the country. When the weightings were light (< 10), large reserve clumps were a common feature in the former homelands. However, once heavier weightings were applied, these large clumps disappeared and the selected areas in the former homelands become highly dispersed. Irrespective of the weighting all conservation targets were met; however, the practicality of the reserve system in the former homelands diminished with heavier weightings. As the weightings increased, the reserve systems became progressively more different to the unweighted scenario (Table 8). According to Spearman’s rank correlation coefficient (ρ) the unweighted scenario was most different to the scenario with a weighting factor of 1000 ($\rho = 0.52$). The scenarios without a weighting and a weighting of 10 generated reserve systems that often selected the same planning units ($\rho = 0.91$). The scenarios with weightings of 100 and 1000 were very similar ($\rho = 0.9$). Therefore, the scenarios without a weighting and a weighting of 10 are most different to the scenarios with weightings of 100 and 1000. When the existing protected areas were removed from the analysis, the similarity between the reserve systems decreased in all instances except for the comparison between the scenarios with weighting factors of 100 and 1000. *Ad hoc* calculations indicated that when heavier weightings are applied, the overall cost of the reserve system increased (Table 9). The reserve system established when the highest weighting was applied had the lowest net value, primarily due to increased opportunity costs. Furthermore, the area: cost and cost: benefit ratios indicate that this reserve system performed the worst in terms of cost effectiveness and economic efficiency. Management costs were lowest for the unweighted scenario, and although the weighted scenarios encountered increased management costs by comparison, management costs did not change substantially between the weighted scenarios.

Table 8: Spearmans rank correlation coefficients (ρ) between weighted opportunity cost scenarios. Weighted_x: x refers to the weighting factor used in the scenario. A weighting factor of 1 is the equivalent of no weighting. Values above the diagonal *exclude* areas locked into the reserve networks; values below the diagonal *include* areas locked into the reserve networks. All values were significant with $p < 0.01$.

	Weighted_1	Weighted_10	Weighted_100	Weighted_1000
Weighted_1	1	0.89	0.65	0.44
Weighted_10	0.91	1	0.84	0.64
Weighted_100	0.7	0.87	1	0.9
Weighted_1000	0.52	0.67	0.9	1

Table 9: Comparison of the weighted scenario best solution reserve networks, excluding existing reserves, in terms of total area (ha), opportunity costs (R), management costs (R), total costs (R), total benefits (R), net value (R), and three ratios: area: cost ratio, area: benefits ratio and cost: benefit ratio. M = millions, B = billions

	Unweighted	Weighted_10	Weighted_100	Weighted_1000
Total area (ha)	22.65 M	24.55 M	23.31 M	23.64 M
Opportunity costs (R)	4.15 B	5.5 B	5.75 B	6.90 B
Management costs (R)	0.21 B	0.31 B	0.30 B	0.30 B
Total costs (R)	4.35 B	5.81 B	6.04 B	7.20 B
Total benefits (R)	11.54 B	13.95 B	11.79 B	11.34 B
Net value (R)	7.39 B	8.45 B	6.04 B	4.44 B
Area: cost ratio	0.0052	0.0042	0.0039	0.0033
Area: benefit ratio	0.0020	0.0018	0.0020	0.0021
Cost: benefit ratio	0.38	0.42	0.51	0.64



Figure 11: The spatial distribution of the 'best solutions' generated by the weighted opportunity cost scenarios. The values of subsistence agriculture and natural resource utilization per planning unit were weighted by factors of 10, 100 and 1000. The legend, scale bar and north arrow apply to all four maps.

Discussion

In a world where land and monetary resources are not readily available, providing an economic case for establishing new protected areas and demonstrating how conservation benefits human well-being are becoming necessary to reinforce arguments for biodiversity protection (Ferraro 2002; Sarkar et al. 2006), and the approach to systematic conservation planning has begun to change accordingly (Balmford et al., 2000; Reyers et al., 2012). Ecosystem services delivered by protected areas provide the link between conservation and human well-being, therefore incorporating ecosystem services into systematic conservation planning has the potential to improve the societal relevance of conservation (Petrosillo et al. 2010). “Efficient” is a term often used to describe the outcomes of the systematic conservation planning process; however, the meaning of “efficiency” in conservation planning has varied. For some, “efficiency” is maximizing the biodiversity gain per unit area and/or per unit cost, while for others it is simply minimizing acquisition, management or opportunity costs (Cameron et al. 2008). From an economic perspective, these two approaches would be said to be achieving cost-effectiveness (cost minimization) as opposed to economic-efficiency (net benefit-maximization). With better understanding of the value of ecosystem services; “efficiency” in conservation planning could come to mean maximizing the net economic benefits of achieving conservation targets. From here on, the term “efficiency” will refer to maximisation of net economic benefits, taking benefits of ecosystem services into account. Whether economic-efficiency should be a defining feature of conservation planning is debatable, however, we are of the opinion that failing to attempt to achieve economic-efficiency will undermine attempts to establish resilient protected areas. Providing an economic case for conservation will, in most cases, increase the support for protected areas establishment, which should theoretically increase the probability of protected areas persisting far into the future. While economic circumstances may become less favourable in the future, an increase in the marginal value of ecosystem service provision from protected areas (as ecosystem service provisioning declines outside of protected areas), may continue to support the original economic case for establishment.

Building on the conclusions drawn by Chan et al. (2011), this study assessed the consequences of explicitly incorporating spatial data on the potential economic costs and benefits of conservation across South Africa. The economic costs of conservation were given as the opportunity cost of foregone agricultural production and natural resource utilization. Economic benefits were considered both in terms of the total value of ecosystem services provided by natural landscapes and the estimated net difference in value between landscapes that are protected vs. unprotected, incorporating expected trajectories of value over a 20 year period.

Study limitations

While the data were the best available at the time, there were several limitations associated with these data. Regarding the commercial agriculture data, data from the 2007 commercial agricultural census would have produced more reliable estimates of the current value of commercial agriculture and a more accurate map of the spatial distribution in current agricultural production. The 2002 commercial agriculture census data were lacking in that no area data relating to income generated from livestock and livestock products were reported. Additionally, net income (as opposed to gross income) could have been used as the economic metric had the expenses associated with commercial agriculture been reported in greater detail. The ecosystem service data set was limited in its resolution and array of ecosystem services captured. Finer resolution ecosystem service data would have contributed toward more accurate cost-benefit data within each planning unit. Including a broader array of ecosystem services, particularly cultural ecosystem services, would have strengthen this study. Similarly, including data relating to ecosystem disservices (disease transmission, crop raiding, etc) would reflect a more accurate picture of the true value of conservation. The use of the ecosystem service data was further limited by the fact that estimating the net benefits of conservation, in terms of the incremental value of ecosystem services delivered from protected areas compared with unprotected areas, relied upon untested assumptions as to how land management influences ecosystem service delivery. Essentially, with the given ecosystem service data, it was not possible to fully capture the influence of feedbacks between people and ecosystems on the economic data. However, even though some ecosystem services may be undervalued or ignored, using the best available data still allows for more informed decisions to be made (Balmford et al. 2011). This study is also somewhat static in that it does not include a temporal dynamics such as land cover change. Where land cover change is a significant and immediate threat to biodiversity, land needs to be conserved sooner rather than later. Therefore, including the probability of land cover change in the vulnerability index used to identify priority areas may be one way in which this temporal dynamic can be incorporated in future. Furthermore, an absence of relevant sociological data may be perceived as a limitation. Despite these uncertainties and limitations, the study still has relevance in that it demonstrates the potential benefits that incorporating spatial cost benefit data into conservation planning has.

Spatial variation in opportunity costs across South Africa

Although this study does not present a full accounting of the opportunity costs associated with biodiversity conservation across South Africa, it is the first study in which the spatial variation in opportunity costs associated with biodiversity conservation across the country has been investigated. Osano (2005) estimated the opportunity costs of biodiversity conservation for the Western Cape, South Africa. In this study, the opportunity costs of conservation were considered to be the sum of the

estimated economic output from current commercial and communal-land agricultural production and natural resource utilization. A full accounting would also include the economic output from forestry as well as some measure of future development potential. As found by Naidoo & Ricketts (2006), the opportunity costs across South Africa were spatially heterogeneous and varied by four orders of magnitude (Figure 5). However, as Arponen et al. (2010) found, the spatial variation in opportunity costs tends to be smaller at a more regional scale (e.g. Karoo and Succulent Karoo). Commercial agricultural production is the primary contributor towards opportunity costs in South Africa. The economic output of commercial agriculture was greatest in areas where both cultivation and livestock production were possible. While the spatial variation in opportunity costs was largely driven by differences in the production potential of the land, this was not the case in former homelands where much of the variation in opportunity costs was due to spatial variation in natural resource utilization.

The more variable the costs, the more important they become in determining efficient conservation plans (Perhans et al. 2008; Polasky 2008). The high degree of spatial variation in opportunity costs across South Africa is a good indication that some measure of economic costs should be included in national conservation planning exercises. Economic costs should also be included in conservation planning exercises when 1) costs vary more than biological value and 2) costs and biological value are positively correlated (Ferraro 2003; Naidoo et al. 2006; Polasky 2008). This study suggests that both of these criteria are likely to be fulfilled for South Africa (Figure 5, Appendix B). Opportunity costs are often more spatially variable than biological data (Polasky 2008), as was the case in this study. However, whether costs vary more than biological value is likely to be dependent on what biodiversity features are targeted in the conservation assessment. This study supports the conclusion by Adams et al. (2010) that opportunity costs can be minimized while achieving biodiversity targets (scenario 2). Furthermore, it demonstrated that explicitly considering opportunity costs can significantly alter priority area networks spatially (Moore et al. 2004; Naidoo & Iwamura 2007).

What are the costs of not explicitly considering ecosystem services

It is increasingly recognised that decision making processes need to take all environmental costs and benefits into account, as omitting some measures will lead to distorted decisions and inefficient outcomes. While conservation assessments are increasingly incorporating opportunity costs, few have considered the impacts of conservation plans on the total value of ecosystem services generated by the planning domain. This study has shown that taking this into account (as a cost layer, rather than as a set of targets) can have further significant impacts on the spatial location and extent of conservation priority areas selected, and also highlights the inefficiency of ignoring these values. Including both economic costs and benefits produced priority area networks that were more efficient in terms of maximizing the economic benefits of the planning domain as a whole. While the *net*

benefit maximization scenario (scenario 3) apparently performed the best, it is important to note that the *conservative net benefit maximization scenario* (scenario 4) is a more appropriate approach and likely to be a better estimation, despite the estimation of additional gains through conservation action being based largely on expert opinion. *Scenario 3* implicitly assumes that all ecosystem services would eventually be lost if the land were not protected, which is not necessarily true. Regulating services are likely to be supplied by unprotected land, albeit as a function of ecosystem health. Although the estimated economic benefits were modest in *scenario 4*, this scenario is still more efficient than *scenarios 1* and *2* in terms of maximizing the net economic benefits per unit cost. Furthermore, it appears that from an efficiency (net economic benefit maximisation) perspective considering economic benefits may be more important when the areas in which benefits exceed the costs are small and localized (*scenario 4*) than when they are large and regional (*scenario 3*). When the areas in which economic benefits exceed costs were large, the priority set from a conservation assessment that explicitly considered only economic benefits (*scenario 2*) was only slightly less efficient than that from *scenario 3*. Valuing ecosystem services is a new field and little is known about how conservation management influences the value of ecosystem services. Polasky et al. (2011) modelled the impact of land-use on ecosystem services and estimated increases and decreases in the value of ecosystem services under conservation. It is generally considered reasonable to assume that conservation management will increase the value of supporting, regulating and cultural services, and decrease the value of provisioning services (Nelson et al. 2009; Petrosillo et al. 2010; Raudsepp-Hearne et al. 2010; Reyers et al. 2012). The set of assumptions made in this study reflect this thinking and were conservative estimates of the additional net benefits that conservation would generate from ecosystem services.

The spatial trade-off between opportunity costs and ecosystem service benefits reflects the pattern found by Naidoo & Ricketts (2006) in that there are areas where ecosystem service benefits exceed opportunity costs. The advantage of spatial cost-benefit analyses is that they make it possible to identify areas where conservation makes more economic sense than it would elsewhere (Naidoo & Ricketts 2006). When economic benefits are explicitly considered, areas that are more economically favourable from a conservation management perspective are prioritized, providing an economic case for establishing protected areas. The opposite is true when economic benefits are not explicitly considered. Therefore, the cost of not explicitly considering the economic benefits is that the implementation potential of priority areas may be compromised, particularly when the economic benefits of competing land-uses are brought into consideration. Although the ecosystem service benefits generated by the priority areas were not orders of magnitude greater than the costs of conserving them (Murdoch et al. 2007; Rands et al. 2010), the results of this study are useful in that they provide an indication of the shortfall that payment for ecosystem service (PES) schemes would not be able to offset (Naidoo & Ricketts 2006). The focus of this study was to assess how ecosystem

system services can be incorporated into conservation planning to motivate for biodiversity conservation. Consequently, the aim was not to prioritise areas of high ecosystem service delivery, as is typically the objective in ecosystem service conservation (Chan et al. 2006). Areas of high ecosystem service delivery are not necessarily congruent with areas of high biodiversity value (Egoh 2009) therefore adequate conservation of both biodiversity and ecosystem services will typically require substantially more land, which is unfeasible for many regions/countries.

This study demonstrates that the efficiency, in terms of net benefit maximization, of a conservation plan can be improved. A conservation assessment designed with implementation in mind would need to explicitly consider additional factors such fragmentation, percentage remaining natural vegetation, spatially variable threats, and other measures of biodiversity. For the purposes of this study, the economic value of land from an agricultural perspective may have acted as a proxy for fragmentation. Cultivation is a major driver of fragmentation in South Africa; therefore, one might expect that fragmentation increases with increases in the value of cultivation. Percentage remaining natural vegetation was accounted for to a degree during the priority area identification process; this may however not be sufficient when the objective is implementation. Despite these limitations, this study is useful for suggesting broad patterns, identifying broad scale priority areas and informing decisions as to what variables should be included in the cost surface of a conservation assessment when economic efficiency is considered to be important end goal. Once broad scale priority areas have been identified, finer scale analyses can be conducted to confirm economically-efficient fine scale priority areas. Despite the decline in spatial variation at local levels, Naidoo & Iwamura (2007) believe that economic costs and benefits still have a role to play in local conservation planning. Spatial heterogeneity in service provision is often a result of differences in ecological and socio-economic conditions at different scales (Chan et al. 2006; De Groot et al. 2010). Finer scale analyses of ecosystem services would be more suited to identifying feedbacks between ecosystem service delivery and land management, feedbacks between different ecosystem services, and assessing how the economic value of a suite of ecosystem services would change under conservation. For instance, the economic value of many regulating (e.g. flood attenuation) and supporting (e.g. pest control) services is intrinsically linked to population density and typically conservation incompatible practices such as agriculture (Singh 2002; Chan et al. 2006). The influence of feedbacks such these would be almost impossible to estimate without some understanding of the localized relationships between ecosystem services and the surrounding socio-economic environment.

Chan et al. (2011) voiced concerns over the spatial cohesion of the reserve system generated by Marxan when ecosystem service benefits were incorporated into the cost surface. They suggested that further experimentation with Marxan's parameters might be a solution to the problem. The reserve systems generated in this study did not conform to this pattern because the Marxan parameters were

adjusted for each scenario to ensure that spatially practical and implementable reserve systems were generated on each occasion. Arponen et al. (2010) raised concerns that reserve systems and priority area networks identified from assessments that incorporated economic variables would fail to meet conservation targets. However, using site-selection software such as Marxan ensures that all targets are met. Marxan's algorithm operates under the constraint that all targets must be met irrespective of the variables are included in the cost-surface. The 'best' reserve systems generated by each scenario in this study met all of the specified conservation targets. The results presented here are in agreement with Carwardine et al. (2010) and Egoh et al. (2010) who did not find any evidence to suggest that biodiversity conservation would be compromised. Unfortunately, due to limited budgets and the need for development, not all of the land selected in the "best" reserve system can be conserved, which means that areas within those reserve systems need to be prioritized for conservation action at the expense of fully meeting all of the specified targets. However, even after the priority area networks had been identified, including economic variables did not compromise the ability of the priority areas to meet conservation targets (scenario 1 compared to scenarios 2, 3 and 4). The fact that the scenarios met <10% of the target for ten of the vegetation bioregions may be due to the way in which the priority areas were identified. Areas that were considered to be highly threatened were prioritized over those that were considered to be less threatened. Many of the vegetation types within these ten bioregions are not currently considered to be endangered or critically endangered, hence their exclusion from the priority set.

The social costs of conservation: is it important to explicitly consider them in the planning process?

Conservationists are well aware of the social implications of establishing protected areas. In many parts of the world establishing protected areas involves relocating people and paying damage compensation when local communities are negatively affected by the animals from protected areas. As mentioned previously, the successful establishment of protected areas is heavily dependent on attitudes and politics. Explicitly incorporating opportunity costs into the prioritization procedure has been proposed as a means for adequately addressing the social costs of conservation (Adams et al. 2010). The short coming of using economic data as a measure of the social costs of conservation is that economic considerations may not always be the dominant motivation for, or against, conserving land. Furthermore, the economic data almost certainly do not capture the complexities that variable political attitudes towards conservation add to this equation. That being said, sociological data that capture these personal and political attitudes are not readily available or suitable for inclusion in analyses such as those in this study. In this study, the opportunity costs comprised economic outputs from commercial agriculture, communal-land agriculture and natural resource utilization. In South Africa, commercial agriculture is typically practiced on privately-owned land, whereas subsistence agriculture and natural resource utilization are typically practiced on communally-owned land in the

former homelands. The opportunity costs may be a reasonable proxy for the social costs of conservation on privately-owned land. Commercial farmers would be adequately compensated for because their farms would be bought at the net present value of the land. However, the same may not be true of the opportunity costs in the former homelands. Where land tenure is communal and population densities are high, determining fair compensation for displacement becomes difficult. It is possible that the opportunity costs of foregone income from subsistence farming and natural resource utilization were an underestimate of the value of the land to the people living in the former homelands. Although the monetary value of the land may be low, the real value of the land lies in its contribution towards livelihoods. Many households rely on government grants, and only 10% of households interviewed in the NIDS (wave 1) reported income from subsistence agriculture. The potential cost of relocating people was not captured in the opportunity costs. Therefore, the opportunity costs in the former homelands were weighted to better capture the social costs of conservation and assess how their inclusion would influence the reserve system. Such weightings have been applied in cost-benefit analyses in the economics literature (Munda et al. 1994; Diakoulaki et al. 1995; Munda 1996, 2004); but have not been applied in conservation planning. Given that the distribution of income across society is likely to vary, integrating this distributional aspect into the cost benefit analysis is important (Munda 1996). Although there are no clear methods for deriving such weights, failing to use any weighting system implies that the existing distribution is optimal and/or the change in income distribution is negligible (Munda 1996). Only if this is a reasonable assumption can unweighted values be used to measure costs and benefits (Munda 1996). When heavier weightings (factors > 100) were applied, the reserve design changed with selected areas in the former homelands shrinking and becoming more dispersed. *Ad hoc* calculations of cost indicate that when weightings increase the reserve system becomes more costly. Opportunity costs increased with each subsequent weighting, while management costs were higher in the weighted scenarios than the unweighted scenario. This suggests that there will be a trade-off between increased compensation costs (when social costs are not considered) and increased reserve costs (when an attempt is made to minimise the social costs). Evaluation of this trade-off will indicate whether it will be more economically beneficial to implement a practical reserve system and relocate more people or implement a slightly less practical reserve system and relocate fewer people. Irrespective of which weighting factor was applied, all conservation targets were met. However, management feasibility of protected areas is an important consideration when designing reserve systems. Protected areas that minimize the social costs of conservation but are dispersed across the landscape may be impossible to manage effectively. Therefore, impractical reserve systems (such as those in the former homelands when heavier weightings were applied) are less likely to be implemented, ultimately compromising biodiversity conservation in those areas. Fine scale planning that explicitly considers social costs may however produce protected areas that are more spatially cohesive.

Implications for systematic conservation planning in South Africa

South Africa is a leading country in terms of designing systematic conservation assessments that support effective implementation of protected areas (Knight et al. 2006). Therefore, research that contributes to improving the implementation potential of conservation assessments should be of value to conservation planners in this country. This study proposes that considering the economic benefits of ecosystem services will change the location of priority areas and improve their economic efficiency, thereby enhancing their implementation potential. In 2009, the NPAES focus areas for conservation action in South Africa were approved (DEAT 2010). These focus areas were identified from two conservation assessments that did not incorporate economic data (DEAT & SANBI 2009). In depth comparisons between the NPAES focus areas and the priority sets from this study were not possible because the focus areas were identified from a far more detailed conservation assessment. However, a visual comparison suggested that the spatial distribution of the NPAES focus areas was not all that different to the priority set identified from *scenario 4* (Figure 10). *Ad hoc* calculations of the opportunity costs and economic benefits indicated that the NPAES focus areas were less efficient (benefit maximization per unit cost and per unit area) than *scenarios 3* and *4*. This suggests that, with minor changes in spatial distribution, the efficiency of the NPAES focus areas could have been improved had economic costs and benefits been incorporated into the conservation assessments. However, the NPAES focus areas did perform better than *scenario 1* in terms of efficiency, suggesting the variables included in the cost-surface of the NPAES conservation assessments were a better proxy for the economic costs and benefits of conservation in South Africa than area is. More generally, the results of this study suggest that if protected areas are to be established within or near the NPAES focus areas, fine-scale conservation assessments that consider ecosystem services and explicitly incorporate both economic costs and benefits into the cost-surface should form the basis for identifying the location of these protected areas. The conclusions, general conservation implications and future research recommendations are discussed in the following chapter.

Chapter 3: Study review – conclusions, conservation implications and future research recommendations

Conclusions

In a first look at the spatial variation in opportunity costs of conservation across South Africa, this study indicated that explicitly incorporating economic costs into conservation assessments for South Africa is important. It also demonstrated that the spatial distribution of conservation priority areas will change when spatially relevant cost benefit data are incorporated into the cost-surface of a systematic conservation assessment. The change in spatial distribution was associated with a gain in economic efficiency (net benefit maximization). Additionally, there was no suggestion that the change in spatial distribution of priority areas compromised biodiversity conservation. Furthermore, this study indicated that the social costs of conservation need to be explicitly considered during the assessment process. An evaluation of the trade-off between increased reserve costs or increased compensation costs is important when trying to achieve economically optimal protected areas. In a country like South Africa, where the type of land tenure is spatially variable, opportunity costs may not fully reflect the social costs of conservation.

Conservation Implications

It is important to bear in mind that the aim of the conservation assessment scenarios presented here was not to prioritize areas of high ecosystem service delivery. The aim was to identify priority areas for meeting conservation targets in areas where conservation management makes more economic sense than it does elsewhere. Therefore, the majority of priority areas were situated in areas where the economic benefits of ecosystem services exceeded, or were not substantially less than, the opportunity costs of conservation. The cost of not incorporating the economic benefits of conservation is that priority areas may have reduced implementation potential, particularly when the economic benefits of competing land-uses are brought to the table. An economic case for conservation that is based upon economic efficiency will strengthen aesthetic and biodiversity motives for conservation. The net value of priority area networks were not positive when the conservative estimates of the additional benefits generated by conservation management were calculated. This information is however useful because it provides an indication of the shortfall that payment for ecosystem services (PES) schemes would not be able to offset. The cost of not appropriately accounting for the social costs of conservation is that excessive compensation costs, which could have been minimized, may be encountered during the implementation phase. The cost of the reserve system changes in accordance with the changes in reserve system design when the social costs of conservation are explicitly

accounted for. It is therefore important to evaluate the trade-off between increases in reserve costs and decreases in compensation costs under different scenarios. Having to relocate large numbers of people has the potential to undermine protected area implementation strategies. However, management feasibility of protected areas is an important consideration when designing reserve systems. Protected areas that minimize the social costs of conservation but are dispersed across the landscape may be impossible to manage effectively. Fine scale planning that explicitly considers social costs may however produce protected areas that are more spatially cohesive.

Future research recommendations

This study highlighted five areas for future research. Within South Africa, collecting economic and social data that are relevant to conservation across the country needs to become a research priority. South Africa needs to establish new protected areas to meet the CBD requirements and the location of these new protected areas should be based upon the outcomes of systematic conservation planning exercises that make explicit use of reliable and relevant economic and social data. Secondly, although there is a need for protected area establishment, alternative solutions to conservation such as biosphere reserves, conservancies and heritage sites should also be considered (Batisse 1982; Cowling 2003) and incorporated into conservation planning exercises. The opportunity costs of these alternative solutions may not be as high, and will undoubtedly promote ecosystem service provision more than unconserved land. It would be worth investigating how the conservation planning approach applied in this study can be expanded to incorporate alternative solutions to conservation. The third and fourth future research recommendations relate to ecosystem services. Ecosystem services need to be valued and mapped at scales that are appropriate for conservation assessments that have implementation as an end goal. The viability of local, regional and national PES schemes also needs to be assessed to determine whether the potential economic benefits of conservation can be realised. Better estimates of the economic benefits of conservation will require research into how land management influences the delivery of a suite of ecosystem services. Furthermore, it is important to begin to understand the feedbacks between ecosystem services themselves. The fifth area of future research requires the conservation planning field to learn from the field of economics. This study suggests that weighting a spatial cost benefit analysis is a potential method for explicitly accounting for the social costs of conservation in systematic conservation planning. The questions that need to be answered are: 1) how can the economic variables be appropriately weighted and 2) will explicitly considering the social costs of conservation improve the implementation potential of a new protected area? Furthermore, introducing time and discounting would produce a more reliable economic analysis.

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Appendix A

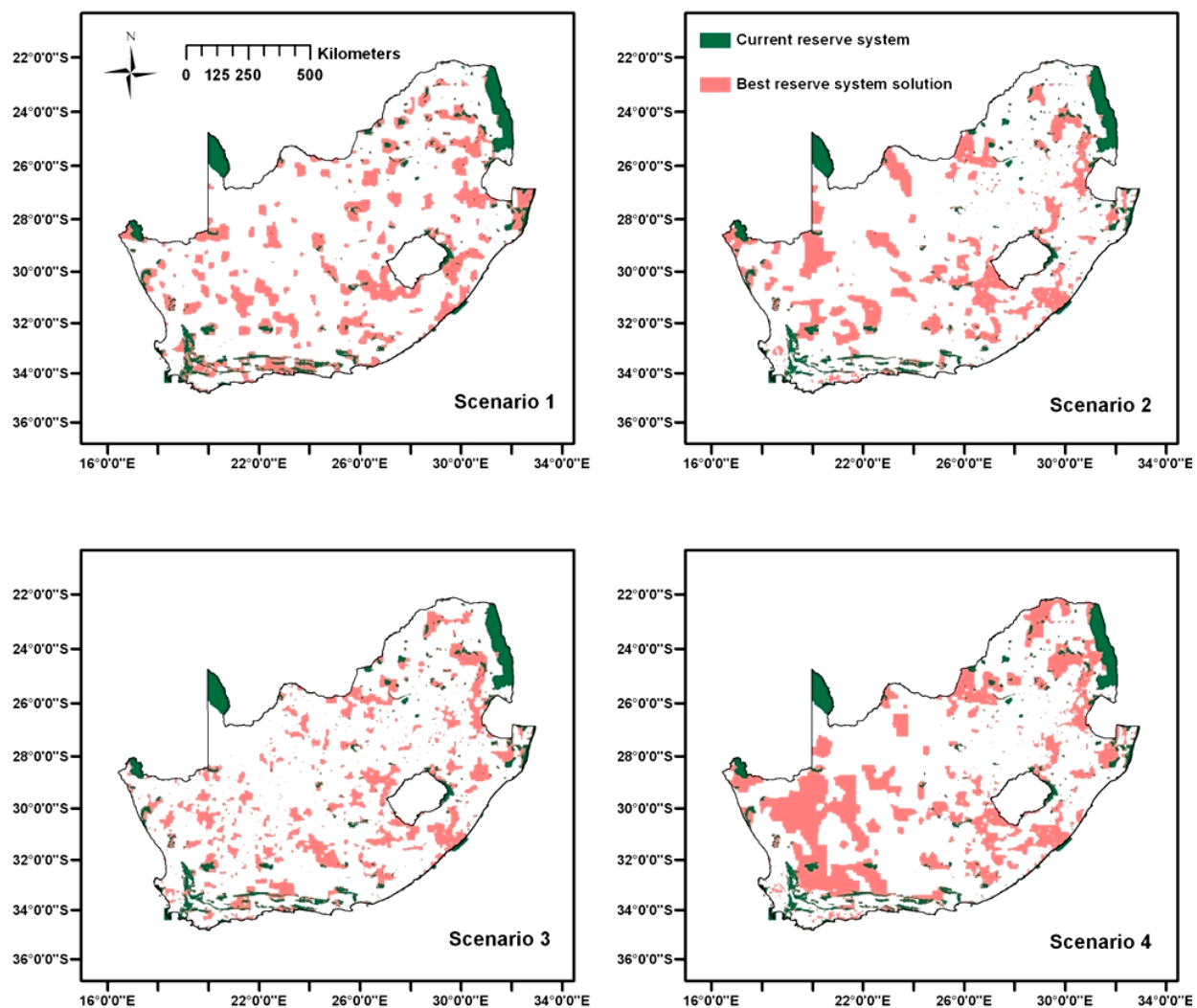


Figure 1: Maps of the "best" reserve systems generated by Marxan for all four scenarios. The best reserve system is indicated in pink; current reserve system is indicated in green.

Appendix B

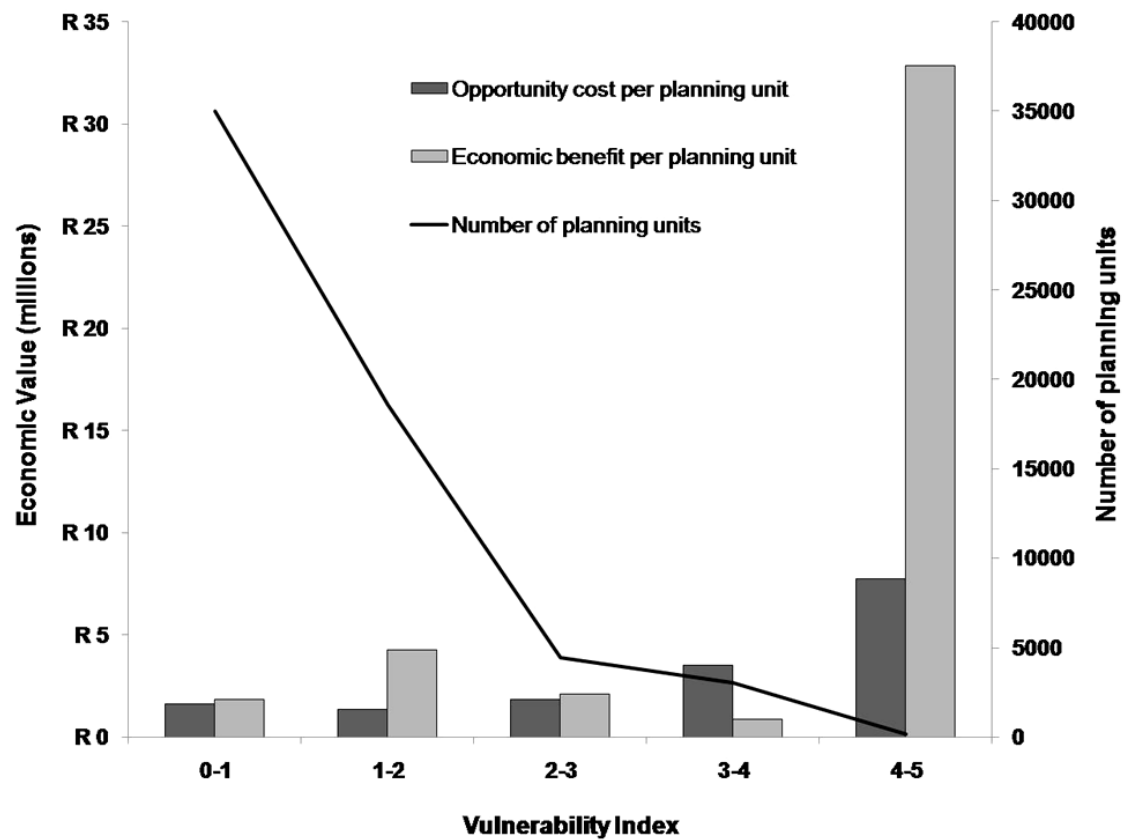


Figure 1: Relationship between the vulnerability of a planning unit and 1) the *average* opportunity cost per planning (dark grey columns) and 2) *average* economic benefit per planning unit (light grey columns). The line graph (secondary axis) indicates the number of planning units that fall into the different vulnerability categories.

Appendix C

Table 1: List of the data used in this study, year of compilation, and data source.

Data	Year	Source
Commercial agriculture	2002	Statistics South Africa, Commercial Agricultural Census 2002, www.statssa.gov.za/agriculture/default.asp
Ecosystem services	2008	Turpie et al. (submitted)
Formal protected areas	2010	South African National Biodiversity Institute (http://bgis.sanbi.org/NBA/project.asp)
Former homelands		Department of Agriculture, Forestry & Fisheries, Directorate: Land use and Soil Management (http://www.agis.agric.za/ags_metadata/Search.jsp?type=category)
Grazing Capacity	2005	Department of Agriculture, Forestry & Fisheries, Directorate: Land use and Soil Management (http://www.agis.agric.za/ags_metadata/Search.jsp?type=category)
Household density	2001	Statistics South Africa, Population Census 2001
Informal protected areas	2010	South African National Biodiversity Institute (http://bgis.sanbi.org/protectedareas/protectedAreas.asp)
Landcover	2009	South African National Biodiversity Institute (http://bgis.sanbi.org/landcover/project.asp)
Municipalities	2005	Council of Scientific and Industrial Research: FUNDISADISK
Magisterial districts	2005	Department of Agriculture, Forestry & Fisheries, Directorate: Land use and Soil Management
South African border		Global Administrative Areas Database (http://www.gadm.org/)
Subsistence agriculture	2008	Southern Africa Labour and Development Research Unit. National Income Dynamics Study 2007-2008, Wave 1 [dataset]. Version 4.1. Cape Town: Southern Africa Labour and Development Research Unit [producer], 2012. Cape Town: DataFirst [distributor], 2012
Urban areas	2012	Council of Scientific and Industrial Research: FUNDISADISK
Vegetation conservation status	2011	South African National Biodiversity Institute (http://bgis.sanbi.org/NBA/project.asp)
Vegetation conservation targets	2011	South African National Biodiversity Institute (http://bgis.sanbi.org/NBA/project.asp)
Vegetation map of South Africa	2006	South African National Biodiversity Institute